WETLAND MITIGATION BANKING IN THE CHICAGO AREA: AN ASSESSMENT OF ECOLOGICAL OUTCOMES AND COMPLIANCE WITH PERFORMANCE STANDARDS

BY

STEPHEN C. TILLMAN

THESIS

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Adviser:

Associate Professor Jeffrey W. Matthews

ABSTRACT

The U.S. Army Corps of Engineers, under Section 404 of the Clean Water Act of 1972, requires that development projects causing negative impacts to wetlands must provide compensation for wetland losses through the wetland mitigation process. The Army Corps prefers that compensation is provided through the purchase of credits from wetland mitigation banks, which are large wetland restoration projects constructed by third-party bank sponsors for the purpose of providing wetland mitigation credits that may be sold for a profit. To evaluate how effectively wetland mitigation banks have achieved the goal of "no net loss" of wetland resources, I have conducted assessments of the regulatory and ecological outcomes of banks in relation to natural wetlands in the Chicago region, which possesses one of the country's most well-developed banking markets. In Chapter 1, I conducted a review of wetland mitigation policy documents and independent research examining banks to provide a definition and thorough description of the practice of wetland mitigation banking. In Chapter 2, I used data from wetland mitigation banks in the Chicago District of the Army Corps to determine how successful banks were at meeting mandatory ecological performance standards, by which the Army Corps evaluates banks at the end of a required monitoring period. In Chapter 3, I used vegetation data that I collected in 2017 from banks that had previously completed their required management and monitoring periods in order to compare the wetlands in banks to natural wetlands in Illinois that were previously sampled by the Illinois Natural History Survey. I made this comparison between banks and natural wetlands using several vegetation-based metrics and using non-metric multidimensional scaling to compare plant species composition. In Chapter 4, I developed a novel simulation modeling approach to determine how effectively banks replace the specific plant species that are lost from the natural wetlands for which banks may be used as

compensation. In Chapter 5, I provided a summary of my primary conclusions from this work. These include the findings that banks typically struggled to meet performance standards limiting dominance by non-native species, while they often met standards related to native species richness and dominance, that the plant communities in banks showed greater ecological quality than those in low-quality, degraded natural wetlands but failed to reach equivalence with high-quality reference wetlands, and that banks typically replaced only about 45% of the native plant species found in impacted natural wetlands which may purchase credits from banks. This work provides new information about the ecological legacy of wetland mitigation banking, which may be used to inform and improve mitigation policy and wetland mitigation bank construction and management.

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CHAPTER 1: GENERAL BACKGROUND

1.1 Introduction

For this thesis I have conducted an evaluation of the regulatory and ecological outcomes of wetland mitigation banks that have been constructed and operated within the Chicago District of the US Army Corps of Engineers (Corps). This project included three separate studies which assessed different features and results from wetland mitigation banking. In this first chapter, I will present a literature review that defines and explains wetland mitigation banking and introduces the task of evaluating banks. In Chapter 2, I will present the results from an assessment of bank compliance with the ecological performance standards that were established by the Corps. For Chapter 3, I collected field vegetation data from wetland mitigation banks that had been completed and finished their required management and monitoring periods, and I used these data to compare the vegetation in banks to that it natural wetlands from the region. In Chapter 4, I present the methods and results from a novel simulation modeling approach that I developed to determine how effectively banks may replace the native plant species present in natural wetlands which the Corps allows to be impacted and compensated for by banks. I conclude with an overall summary of my results in Chapter 5.

1.2 Literature Review

Wetland mitigation in the United States

The federal government of the United States regulates impacts to wetlands primarily through policy that is based upon Section 404 of the Clean Water Act (CWA) of 1972. The task of administering this legislation belongs to the Corps and the US Environmental Protection Agency (EPA). An objective of the CWA is to “restore and maintain the chemical, physical and biological integrity of the Nation’s waters, including wetlands” (Corps and EPA 1990). As part of this objective, the Corps has established the specific goal of achieving "no overall net loss of values and functions" of wetlands (Corps and EPA 1990). In service of this goal, the Corps attempts to avoid adverse impacts to aquatic resources, and to offset impacts that are deemed unavoidable. All development and construction projects that impact wetlands must be approved and permitted by the Corps. This process of preventing and then offsetting wetland impacts is called mitigation and involves three steps defined in a 1990 Memorandum of Agreement between the EPA and the Corps (Corps and EPA 1990). First, permitted projects must take all appropriate and practicable steps to avoid adverse impacts to wetlands. Next, any unavoidable wetland impacts must be minimized. Finally, the remaining adverse impacts to aquatic resources must be compensated for by the generation of new aquatic resources and values, an action referred to as compensatory mitigation. Compensation may be provided at the same location at which adverse impacts occur (on-site compensation) or compensation may be implemented at a different site (off-site compensation).

Compensation for unavoidable impacts to aquatic resources is conducted through the exchange of mitigation credits, which are based on the specific ecological characteristics of impacted and compensation sites (Corps and EPA 2008). Credits are most often defined in acres. The Corps states that the amount of compensatory mitigation credits that are required should be adequate to replace the lost aquatic resource functions. It is preferred that this amount be determined using functional or condition assessment methods to measure the resources that will be impacted, but when these methods are not used, a minimum one-to-one mitigation ratio is required between the acres impacted and the number of credits required as compensation (Corps and EPA 2008). Regional Corps Districts may set the minimum mitigation ratio at more than one-to-one, requiring the permittee to provide more compensation credits than the number of acres that have been impacted (US Army Corps of Engineers Chicago District 2009). The Corps may change the mitigation ratio depending on the method of compensatory mitigation that is used, the likelihood of success of compensation, differences between the functions impacted and those that will be produced by compensation, temporal losses of aquatic resources that result when impacts occur before compensation is implemented, and the distance between the impact and compensation site (Corps and EPA 2008).

There are four methods by which compensatory wetland mitigation projects generate aquatic resource values to offset those lost at impacted wetlands: restoration, creation (i.e. establishment), enhancement, and preservation (IWR 2015). Restoration activities return wetland function to a degraded non-wetland site that previously existed as a wetland. Creation produces a wetland at an upland site that did not exist historically as a wetland. Enhancement activities improve wetland resource functions at an existing wetland site. Preservation removes a threat to, or prevents the decline of, an existing wetland, usually by providing legal or physical protection to the site. The Corps has determined restoration to be the preferred method of compensatory mitigation because it provides the greatest likelihood of success, it results in an increase in wetland area (unlike wetland enhancement or preservation), and it reduces the impact to uplands compared to wetland creation (Corps and EPA 2008).

There are three mechanisms by which a permittee may provide compensatory wetland mitigation: permittee-responsible mitigation, in-lieu fee mitigation, and mitigation banks (IWR 2015). When using permittee-responsible mitigation, permittees themselves retain responsibility for providing successful compensation, though they may hire environmental contractors to perform the work. Permittee-responsible compensation projects are conducted by just one permittee to satisfy that permittee's compensation responsibilities. In-lieu fee projects and mitigation banks are generally implemented by third parties, referred to as project sponsors, which perform off-site compensation activities designed to offset permitted wetland impacts for multiple permittees. During in-lieu fee mitigation, permittees pay an in-lieu fee sponsor, which must be a government agency or non-profit conservation organization, to satisfy their compensation requirements. The sponsor pools the funds collected from multiple impacts and permittees, then uses them to restore wetlands in an in-lieu fee project. In contrast, mitigation banks are usually sponsored by private companies, which provide the initial investment to conduct wetland restoration that will generate mitigation credits, which can then be sold to permittees to satisfy their responsibility to compensate for wetland impacts. Before credits can be sold, the Corps must determine that the credit-generating restoration at banks has been completed and successfully reached certain milestones. In this way, mitigation banks generally provide wetland compensation before permitted impacts to wetlands occur, while permittee-responsible and in-lieu fee projects create a lag time between wetland impacts and the execution of compensation. This is not always the case however, as banks may be permitted to sell some credits before a compensation project has been implemented, or before it has been judged to be successful (Corps and EPA 2008, IRT 2008).

Wetland mitigation banks

The establishment and management of banks is regulated by an Interagency Review Team (IRT) that includes representatives from the Corps, the EPA, and other federal and local regulatory and resource agencies (Corps and EPA 2008). Bank sponsors must have a mitigation and management plan approved by the IRT before a bank can be constructed. This plan must include a description of the method and amount of compensation that will be provided, the number of credits that will be generated, and provisions for the immediate and long-term management and protection of the site. If the IRT approves a mitigation bank, the conditions for its creation and management are formalized in a document called the mitigation banking instrument (Corps and EPA 2008). The number of credits that the IRT determines may be produced by a mitigation bank depends on the type and acreage of wetlands or upland buffers that are generated at the bank site. A single bank may use more than one of the four methods of mitigation (restoration, creation, enhancement, and preservation) to produce credits, but the rate of credit production may differ depending on which method is used (Corps and EPA 2008). The Corps favors restored or created wetlands because these methods create new wetland area, so these may be credited at a 1:1 ratio of acres of restoration to credits. The Corps generally grants fewer credits per acre for wetland enhancement or preservation. Mitigation banks can also gain a limited number of credits by establishing upland buffers around wetlands.

A fundamental characteristic of wetland mitigation banks is that they facilitate the spatial redistribution and consolidation of wetlands by compensating for multiple spatially distributed wetland impacts at one mitigation site (Brown and Lant 1999). This is implicit in the concept of no net loss of wetland resources. Rather than requiring all wetland impacts be offset by on-site compensation, which would constitute no absolute loss of wetlands, mitigation banking allows for compensation to be redistributed geographically while ensuring that the net quantity of wetland resources is maintained (Brown and Lant 1999). Mitigation banks do have limitations on the geographic area within which they can sell credits to offset wetland impacts; this is the bank's service area (Corps and EPA 2008). The extent of the service area varies between mitigation banks and is determined by the Corps. The Corps typically requires that compensation for wetland impacts be implemented in the same watershed as the impact site. Bank service areas must be sized so that the aquatic resources provided by a bank can appropriately compensate for adverse impacts to wetlands across the entire service area (Corps and EPA 2008).

Once the bank has been constructed, it must be monitored so that the IRT can evaluate the site's compliance with specific hydrologic and ecological performance standards. Performance standards are designed to objectively evaluate if a mitigation bank is developing the wetland resource types, wetland functions, and wetland acreage that are prescribed by its banking instrument (Corps and EPA 2008). The Chicago District of the Corps states that the purpose of site monitoring and performance standards is to "ensure that mitigation banks create aquatic resources…which compare favorably with moderate to high quality natural aquatic resources/wetlands with respect to diversity, abundance and distribution of plant species, and also to ensure that the created aquatic resources/wetlands exhibit the hydrologic regimes of natural wetlands" (IRT 2008). Each mitigation bank may have its own specific performance standards and thresholds, but they are often similar or identical between banks, and share an emphasis on vegetation-based attributes and wetland hydrology (Matthews and Endress 2008, Reiss et al. 2009). Standards should be based on ecological attributes that are verifiable, and can be practically measured (Streever 1999, Corps and EPA 2008).

A mitigation bank's compliance with its performance standards and the IRT's decision to approve bank credits for sale are determined from monitoring data that must be collected from the bank according to its banking instrument. Much of a bank's credits may be released for sale prior to the end of the required monitoring period if the bank hits appropriate management and performance milestones, but a bank cannot receive full credit release until it has met all performance standards at the completion of its required monitoring (Corps and EPA 2008). The data collection methods, parameters to be sampled and monitoring schedule vary by bank and are dependent on site-specific characteristics and performance standards. The monitoring period must last for a minimum of five years (Corps and EPA 2008), though a number of researchers have argued that this period of time is insufficient to evaluate the ecological outcomes of mitigation wetlands (Spieles et al. 2006, Aronson and Galatowitsch 2008, Matthews et al. 2009, Stefanik and Mitsch 2012). The IRT determines that a mitigation bank has successfully produced all of the wetland resources for which it was designed when it reaches the end of its monitoring period and has met all of its performance standards and long-term management obligations (IRT 2008). The IRT will then release all of the bank's remaining credits, so that they may be sold to wetland impact permittees. If the IRT determines that a mitigation bank is not meeting its performance standards, it may extend the original monitoring period and revise the monitoring requirements, and the bank sponsor may be required to conduct remediation and/or adaptive management on the site (Corps and EPA 2008).

Entrepreneurial wetland mitigation banking first developed in the early 1990's (Hough and Robertson 2009) and in 2008 the Corps and EPA officially identified mitigation banks as the preferred mechanism for compensating for wetland impacts (Corps and EPA 2008). As justification, the Corps listed several advantages of mitigation banking. Banks must provide a mitigation plan and financial assurances before the bank is constructed, and the Corps expects that this reduces the risk and uncertainty that banks will fail to produce adequate wetland resources. Banks generally must restore, create, or enhance wetlands before their credits can be sold, which removes the delay between impacts to wetlands and the provision of appropriate compensation (Corps and EPA 2008). The Corps also indicates that banks facilitate compensatory mitigation projects that are larger and more ecologically valuable, include more rigorous scientific planning, and attract a greater investment of financial resources than other mitigation mechanisms (Corps and EPA 2008). Banks also improve the cost-effectiveness of wetland mitigation by taking advantage of economies of scale for bank construction and management (Brown and Lant 1999). The increasing preference for mitigation banks has coincided with an increase in their use. Historically, permittee-responsible mitigation was used more often than banks, but by 2014 banks were the most frequently used mitigation mechanism (IWR 2015), and the percentage of wetland mitigation that uses banks continued to increase from 2014 to 2017 (Hough and Harrington 2019). As mitigation banks have been used more frequently to compensate for wetland impacts, it has become important to evaluate how well they are replacing impacted wetland resources.

Comparing mitigation wetlands to natural reference wetlands

In order to define goals for compensatory mitigation wetlands (including permittee-responsible, in-lieu fee, and mitigation bank projects), and to judge if they are meeting their goals, it must be determined how mitigation wetlands should be evaluated. Many have suggested that mitigation projects should be assessed in comparison to natural reference sites of the same wetland type (Kentula et al. 1992, Brinson and Rheinhardt 1996, Reiss et al. 2007, Fennessy et al. 2013) and that mitigation wetlands should be based on the structural and functional characteristics, hydrogeomorphic class, and vegetation type of reference wetlands (Brooks et al. 2005). To compare mitigation projects to reference wetlands, it is useful to use a population of reference wetlands for comparison, rather than single sites, so that the natural variability present among reference wetlands is considered (Kentula et al. 1992). When possible, it may be helpful to evaluate mitigation wetlands relative both to ambient natural wetlands that include the full range of human disturbance present in a landscape, and to “reference-standard” wetlands that represent the least disturbed conditions (Fennessy et al. 2013).

Ecological outcomes of wetland mitigation banks

While studies of compensatory wetland mitigation have usually focused on permittee-responsible mitigation, rather than mitigation banks (Morgan and Hough 2015), some studies have specifically examined the regulatory and ecological performance of banks. The results have been variable; as some banks are able to produce wetland resources of reasonable quality, while others fail even to produce sufficient wetland area. An early national survey found that the use of mitigation banking had created an overall net loss in wetland area (Brown and Lant 1999); however, more recent studies suggest an improvement in this regard (Robertson and Hayden 2008).

Evaluations of the ecological condition of the plant communities in mitigation banks have produced mixed results concerning the ability of banks to produce wetland plant communities that are equivalent with those in natural wetlands. Spieles et al. (2006) found that plant communities were similar between banks and reference wetlands ten years after bank construction was completed. Stefanik and Mitsch (2012) found that banks had, on average, species richness and floristic quality measures lower than those in natural wetlands, but that some individual banks did appear comparable to natural wetlands by these metrics. The level of ecological succession and community establishment in banks, as indicated by the composition of functional plant groups, was lower than in reference wetlands, but improving trends in older bank sites indicated that the vegetation communities in banks may still be maturing toward reference conditions. This study also showed that the structural characteristics of vegetation communities in younger mitigation banks were more similar to reference wetlands than were those in older banks, suggesting that the design and construction techniques used for these banks may have improved over time (Stefanik and Mitsch 2012). A study of wetland mitigation in Ohio found that mitigation bank wetlands were equivalent with natural sites of fair to good quality (PGE and MBI 2012). A greater number of mitigation banks sampled in 2011 met performance standards measuring biotic integrity than did banks sampled from 2001-2004, indicating a potential improvement in the quality of mitigation bank wetlands (PGE and MBI 2012). Reiss et al. (2007) found evidence for poor ecological and functional performance in banks, reporting that few, if any, mitigation bank wetlands in Florida produced functional assessment scores equivalent to reference standards. Spieles (2005) observed that vegetation trends in mitigation banks in the first five years following construction were unclear and sometimes erratic, suggesting that a monitoring period of longer than five years is necessary to assess the vegetation conditions in mitigation bank wetlands following this initial period of self-organization.

Ecological conditions in other restored wetlands

In addition to work that has focused just on wetland mitigation banks, many studies have evaluated the ability of restored wetlands in general to produce plant communities like those in natural wetlands. A number of authors, using a variety of vegetation-based metrics, have found the ecological condition of restored wetlands to be variable, and have identified a number of problems that often prevent restorations from achieving equivalence with natural reference wetlands. Studies that have used measures of plant species composition to evaluate trends in restored wetland sites have generally found that restored wetlands may become more similar to each other, less similar to high-quality natural wetlands, and more similar to low-quality natural wetlands over time (Aronson and Galatowitsch 2008, Matthews and Spyreas 2010). Species richness in restored wetlands can fluctuate over time, sometimes approaching the level observed in reference wetlands (Gutrich et al. 2009, Matthews et al. 2009, Hopple and Craft 2013), but in other cases falling well short of species richness in reference sites (Campbell et al. 2002, Gutrich et al. 2009). The metrics chosen to evaluate wetland restorations require careful consideration, as similar measures of floristic quality with subtle differences may suggest very different results about the condition of restored wetlands relative to natural wetland sites (Matthews et al. 2009, Hopple and Craft 2013). A common theme in many studies of wetland restoration is that the ecological integrity of restored sites is often compromised by the presence of invasive plant species (Aronson and Galatowitsch 2008, Gutrich et al. 2009, Matthews et al. 2009, Matthews and Spyreas 2010).

1.3 Conclusion

Regulatory agencies have prescribed the use of wetland mitigation banks as the preferred method of wetland compensation. If banks are able to restore wetlands that are equivalent with natural wetlands, then mitigation banking can be viewed as a useful tool that may allow the achievement of no net loss of wetland resources, even as economic development continues. However, if banks fail to meet their performance standards or produce wetlands similar to those in natural wetlands, then regulatory agencies' confidence in banking will lead to further loss of wetland resources. There is some evidence that restored wetlands, and the wetlands in mitigation banks specifically, may be able to attain a level of ecological quality similar to natural reference wetlands, but this outcome is not guaranteed, and many studies have shown that restorations fail to reach this goal. More work is needed to assess how the plant communities in banks perform relative to their performance standards and to the natural wetlands for which they are used as compensation, particularly in the years following the conclusion of their required monitoring periods. Studies that have previously evaluated banks have typically used general measures of vegetation structure and quality, but to understand if banks can adequately replace the wetland resources in natural wetlands it will also be helpful to evaluate the ability of banks to replace the specific wetland components (i.e. plant species) present in the natural wetlands to which they provide credits. To address these research needs, in this thesis I have evaluated the regulatory performance of mitigation banks, compared the plant communities in mitigation banks to those in reference wetlands for banks of varying ages, and determined if banks are replacing impacted wetland plant species so that no net loss of species is achieved. I have completed this work by evaluating wetland mitigation banks that were owned and operated by private banks sponsors within the Chicago District of the Corps, which includes Cook, DuPage, Kane, Lake, McHenry, and Will counties in northeastern Illinois (Figure 1.1). The Chicago District provides an excellent study area for wetland mitigation banking, as it possesses one of the oldest and most well-developed banking markets in the country.

1.4 Figure

Figure 1.1. Map of Illinois, highlighting the Chicago District of the Army Corps, which includes six counties in the northeastern corner of the state (Cook, DuPage, Kane, Lake, McHenry, Will).

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CHAPTER 2: COMPLIANCE WITH REGULATORY PERFORMANCE STANDARDS DURING THE FINAL YEAR OF MONITORING IN WETLAND MITIGATION BANKS

2.1 Introduction

Wetland mitigation banking is an environmental offsetting mechanism that has been developed and adopted by the U.S. Army Corps of Engineers (Corps) and U.S. Environmental Protection Agency (EPA) as a tool for conserving wetland resources. Under current policies, which are originally based on Section 404 of the Clean Water Act of 1972, any party wishing to conduct land development that will negatively affect jurisdictional wetlands must first receive a permit from the Corps. Permittees must attempt to design their development projects to avoid and minimize negative impacts to wetlands, but if some impacts are found to be unavoidable, then the permittee will be required to provide compensatory mitigation (Corps and EPA 1990). Compensatory mitigation consists of actions taken to produce or improve wetland resources that will offset those lost to the permitted development, so that there will be “no overall net loss of values and functions” of wetlands (Corps and EPA 1990). Compensatory mitigation is achieved through projects designed to restore, create, enhance, or preserve wetlands (IWR 2015).

In order to satisfy their compensatory mitigation requirements, permittees may sometimes choose to independently fund and construct a wetland mitigation project in a process known as permittee-responsible mitigation, but in many cases, wetland mitigation banking presents an appealing alternative (Corps and EPA 2008). In the process of wetland mitigation banking, a third-party individual or company, designated as the bank sponsor, constructs a large wetland restoration project known as a wetland mitigation bank to generate wetland resources that may be used as compensatory mitigation. If regulatory agencies determine that a bank has successfully produced these resources, then they will release to the bank sponsor an amount of wetland credits that is proportionate to the amount of wetland resources produced by the bank.

The bank sponsor may then sell these credits for a profit to permittees, thereby satisfying the permittees' responsibility to provide compensatory mitigation. Permittee-responsible mitigation was the first, and historically the most frequently used, mechanism for mitigation, but in 2008, the Corps and EPA identified wetland mitigation banking as the preferred method of compensating for permitted impacts to wetlands (Corps and EPA 2008). This preference for mitigation banks has coincided with an increase in their use. The number of mitigation banks has grown since 1995, with an increase in growth after 2008, and since 2014, permittees have used banks more than any other mitigation mechanism (IWR 2015, Hough and Harrington 2019).

Wetland mitigation banks are regulated in each individual Corps district by an Interagency Review Team (IRT) of federal, tribal, state, or local agencies (Corps and EPA 2008). In the Chicago District of the Corps, for example, the Corps and EPA are joined on the IRT by the Chicago Field Office of the U.S. Fish and Wildlife Service (IRT 2017). The IRT is responsible for approving proposals to construct mitigation banks, regulating the construction and operation of banks, and providing final approval and credit release for banks (Corps and EPA 2008). In performing these tasks, the IRT must ensure that wetland mitigation banks produce wetlands of satisfactory quality to offset the wetland losses for which they are used as compensation. To do this, the IRT uses specific hydrologic and ecological performance standards to evaluate banks. Performance standards are designed to objectively and quantitatively determine if a wetland mitigation bank has developed the wetland resource types, wetland functions, and wetland acreage that are prescribed in its banking instrument, a project-specific mitigation plan approved by the IRT (Corps and EPA 2008). In the Chicago District of the Corps, the stated purpose of site monitoring and performance standards is to "ensure that mitigation banks create aquatic resources…which compare favorably with moderate to high quality natural aquatic resources/wetlands with respect to diversity, abundance and distribution of plant species, and also to ensure that the created aquatic resources/wetlands exhibit the hydrologic regimes of natural wetlands" (IRT 2008). Each mitigation bank may have its own specific performance standards and thresholds, but they are often similar or identical between banks within a Corps district, and typically share an emphasis on vegetation-based attributes and wetland hydrology. Standards should be based on ecological attributes that are verifiable and can be practically measured (Streever 1999, Corps and EPA 2008).

The specific performance standards that are chosen for mitigation sites determine which wetland resources, and which measures of those resources, are used to evaluate the mitigation project's compliance with its permit. Wetland mitigation policy is particularly concerned with the conservation of wetland functions (Corps and EPA 1990), but rather than directly measuring wetland function, measures of wetland structure are often used as indicators because they are easier and cheaper to measure, and because some measures of structure may serve to indicate changes in function over time (Kentula et al. 1992). However, some have questioned the assumption that measures of wetland structure are an accurate indicator of function, arguing that this assumption is largely untested and theoretically unsound (Cole 2002). The most common performance standards are based on the vegetation present in mitigation wetlands and may include requirements for vegetation cover and species richness, dominance by native perennial plant species, and limitations on non-native or weedy species (Streever 1999, Environmental Law Institute 2002, Matthews and Endress 2008, Reiss et al. 2009). In addition to vegetation-based standards, a common requirement is that the wetland areas in mitigation projects must meet the vegetation, soil, and hydrologic requirements necessary to be defined legally as jurisdictional wetlands (Environmental Laboratory 1987, Streever 1999, Matthews and Endress

2008). Additional standards related to hydrology are also used for mitigation wetlands (Environmental Law Institute 2002, Reiss et al. 2009). Other, less common, performance standards may be related specifically to water quality, hydric soils, and wildlife habitat and use (Environmental Law Institute 2002).

Regulatory agencies require bank sponsors to collect monitoring data that are used to determine a bank's compliance with its performance standards. The data collection methods, parameters to be sampled and monitoring schedule vary by bank and are dependent on site-specific characteristics and performance standards. According to regulatory documents, the monitoring period must last for a minimum of five years (Corps and EPA 2008). At the end of this monitoring period, the IRT reviews the data to determine if the bank has successfully met its performance standards and produced all the wetland resources for which it was designed (IRT 2008). While much of a bank's mitigation credits may be released as the site hits certain construction and performance milestones before the end of its monitoring period, the IRT will not release all of a bank's credits until it has met its performance standards. If the IRT determines that a mitigation bank is not meeting its performance standards, it may extend and revise the monitoring requirements until the bank meets its standards, and the bank sponsor may be required to conduct remediation and/or adaptive management on the site (Corps and EPA 2008).

Several studies that have examined compliance with ecological performance standards in permittee-responsible wetland mitigation projects have found that these projects were seldom able to meet all their performance standards (Morgan and Roberts 2003, Matthews and Endress 2008, Van den Bosh and Matthews 2017). In comparison, some have found that wetland mitigation banks, specifically, have previously achieved relatively high compliance with their permit conditions, but these studies have been able to sample only a small number of banks (Spieles et al. 2006) or assessed performance criteria that were based mostly on task completion rather than measurable, ecological performance criteria (Reiss et al. 2009). In this study, I will evaluate compliance from more than 20 wetland mitigation banks by thoroughly assessing whether banks have been able to meet their specific, measurable, ecological performance standards. By evaluating bank performance, my study will also help to identify which standards have been most difficult for banks to achieve. I will also examine if the values of banks' vegetation-based performance metrics changed over time in banks, to assess the appropriateness of evaluating a bank's performance after five years. The objectives of this study are to 1) determine the performance of wetland mitigation banks in their final year of monitoring relative to their vegetation-based regulatory performance standards, and to identify for which performance standards were compliance by banks the lowest, and 2) to assess if the vegetation metric scores on which performance standards are based changed during the banks' five-year monitoring period.

2.2 Methods

Collecting data from monitoring reports

I conducted this study using wetland mitigation banks regulated by the Chicago District of the Corps, which includes Cook, DuPage, Kane, Lake, McHenry, and Will counties in northeastern Illinois. This study system is particularly useful for mitigation bank research because the Chicago District has a relatively large and well-developed wetland mitigation banking market, which includes some of the oldest private mitigation banks in the country. Information about banks, including a list of bank sites, is publicly available at the Corps' online database, the Regulatory In-lieu Fee and Bank Information Tracking System (RIBITS) (Corps

2020). In this study, I included every Chicago District bank listed on RIBITS which had received final approval and credit release from the IRT as of January 2018, and for which I was able to obtain sufficient monitoring data. These banks were constructed and managed by several different bank sponsors, most of which were privately owned ecological restoration firms. Some banks were separated spatially into multiple phases, which were typically adjacent to each other on the same property but may have been constructed in different years. My dataset from the Chicago District contained seven such banks with two phases each. The IRT required bank sponsors to collect monitoring data and meet performance standards for each separate phase. Following this approach, I treated each phase within a bank as an independent site, which gave me a total of 27 different bank sites eligible for inclusion in my study. Initial construction on the oldest banks in my study occurred in 1997, while the youngest bank was constructed in 2013. Bank size ranged from 19.22 to 94.02 acres (mean = 52.85).

To evaluate the achievement of performance standards in banks, I used data collected from annual monitoring reports that bank sponsors were required to submit to the IRT. I obtained these reports from RIBITS, the U.S. EPA Region 5 Main Office in Chicago, and the bank sponsors themselves. Annual mitigation bank monitoring reports included a summary of the project, a description of construction and management activities, raw data from ecological monitoring, and a summary of the site condition describing whether it had met performance standards. The bank sponsor usually provided tables to summarize the monitoring data and show if the bank had met each of its individual performance standards, but I found some variation in the way that performance standards were applied and reported between different bank sponsors. To ensure that I assessed performance consistently for all bank sites, I chose to use the raw monitoring data from each bank and to apply the performance standards to these data myself, rather than relying on the bank sponsors' summaries of performance.

Monitoring reports from each bank included vegetation quadrat data collected from individual sampling units, which were usually permanent transects, within the bank. The quadrat data reported for each bank included the name, frequency, and cover of each plant species recorded within sampling quadrats. There was variation both between banks and between monitoring years at a single bank in transect arrangement, the number of transects (between 1 and 20), the number of quadrats (between 10 and 532), quadrat shape (square or circular), and quadrat size (0.25-m^2 or 1-m^2). Monitoring reports typically included additional data such as plant species lists generated from timed meander searches of each sampling unit, a plant species inventory for the entire site, and several different forms of hydrology data. The quality and format of timed meander search and hydrology data varied among banks, so I limited my analysis to vegetation quadrat data. This omission of timed meander search data likely affected my analysis of certain performance standards, which were designed to include these data; this will be discussed for these standards later in this paper.

For some banks and in some monitoring years, quadrat sampling was conducted twice during the growing season. Since the number and timing of sampling visits was not consistent across banks, I chose to include quadrat data from just one sampling visit per year at each bank. When monitoring data from a bank were reported from two sampling visits in a year, I chose to include whichever of these visits fell during or closest to the peak of the growing season, between the months of June and August. If data from only one sampling visit were included in a monitoring report, and this visit fell outside of my June to August window, I still chose to include these data in my analysis (in these cases, monitoring took place during May, September, or October).

Selecting monitoring years for data analysis

I used two different approaches of evaluating monitoring data to complete my research objectives: 1) an assessment of bank performance in the final year of monitoring and 2) a temporal analysis of vegetation metrics in two different monitoring years to determine if bank condition relative to these performance metrics changed through time. According to regulatory documents, bank sponsors were required to submit annual monitoring reports for each year until a bank met all its performance standards and received final credit release from the IRT (IRT 2008), but for most banks I found that monitoring reports for some years of operation were not available or were never produced. As a result, I had to allow some flexibility in the monitoring years I selected for analysis.

Objective 1: To evaluate the performance of banks in their final monitoring year I analyzed the last monitoring report available for each bank. This final monitoring data would ideally come from the last growing season before the bank received final approval and credit release and this was the case for eight banks; however, since I was not able to obtain a monitoring report from each year of operation at some banks, I also included nine banks from which the final monitoring data were collected one year before final credit release occurred. I also chose to include one bank from which the final monitoring data I obtained (from the fifth growing season) were collected two years before the seventh and final growing season. I did this after determining that the data from the fifth growing season represented a vegetation community that was relatively stable, and unlikely to change dramatically between the fifth and seventh growing seasons. I excluded two banks from which the latest monitoring data I obtained were not collected within two years of final approval and credit release, since these data would not have accurately represented the data used for the final performance standard evaluation in these banks. There were four banks which I knew were granted final credit release, but I could not determine when the final growing season occurred; however, I did obtain monitoring data at all of these banks from between the fourth and seventh years of operation. I chose to include these banks in my analysis even though I could not be certain that my data represented the final monitoring year. I justified this decision by making the assumption that the data I obtained were likely collected within one or two years of the last growing season, since four to seven was a typical age at which banks received final site evaluation and approval. Following these protocols, the final year monitoring data I assessed represented banks with an average age of 4.2 years and a median age of 3 years since the first growing season. The two youngest banks for which I assessed final monitoring data were only 1 year old, while the oldest bank was 11 years old.

The regulatory documents governing the banks included in my study state that banks must be managed and monitored for at least five years before they can be granted final credit release, but that the monitoring period may be shortened if the bank meets its performance standards in less than five years (Corps and EPA 2008, IRT 2008). 17 of the 26 banks in my study were managed and monitored for at least five growing seasons, but I also found that some of the bank phases in my study received final credit release as early as their second growing season since site construction. While the development and condition of vegetation in a two-year-old restored wetland could be expected to differ from that in a five-year-old restoration site (Matthews et al. 2009), I still chose to include two banks that received credit release as early as their second year because these sites were determined by the IRT to have met their performance standards, and therefore still gave me the opportunity to evaluate banks in their final year of monitoring.

Objective 2: To complete my temporal analysis of the change in vegetation metrics in banks throughout the monitoring period, I required two years of monitoring data from each bank. I decided that analyzing monitoring data from consecutive years would not allow enough time for significant vegetation development to occur, so I chose to include only those banks for which I could obtain paired monitoring data with a two- or three-year gap between monitoring visits. When there was more than one possible choice of paired monitoring years at a single bank, I chose the following priority order to select just one set of paired data from each bank: years three and five, years four and six, years two and four, and years three and six. I excluded two banks from this analysis because I possessed paired data only from years one and three, and I decided that bank vegetation in the first growing season after site construction would not have developed sufficiently for this quantitative assessment.

Performance standards analysis

The performance standards that mitigation banks in the Chicago District must meet are determined by the Chicago Region IRT and have been updated over time in several Interagency Coordination Agreements (ICAs) (IRT 1997, IRT 2008, IRT 2017). The performance standards applied by the IRT to each mitigation bank reflect the version of the ICA under which the bank was permitted. Of the sites considered in my study, three were operated under policies that preceded the 1997 ICA, twenty-two followed the 1997 ICA, and two followed the 2008 ICA. I assessed bank performance at all sites relative to several standards from the 1997 and 2008 ICA agreements, as these regulatory agreements were most relevant to the management and evaluation of the banks in my study.

In the annual monitoring reports, bank sponsors usually identified individual sampling units within banks as belonging to one of several different plant community types. For some performance standards assigned to the banks included in my study, the threshold banks were required to meet to achieve the standard were different for each of three plant community types: marsh, sedge meadow/wet prairie (hereafter referred to as wet meadow), and mesic prairie buffer (hereafter referred to as buffer). Based on the habitat descriptions included in the monitoring data, I assigned each sampling unit from the banks into one of these three community types. I assigned sampling units that were described in monitoring reports as marsh, hemi-marsh, emergent, aquatic-emergent, and riverine backwater into the marsh category. I assigned units described as sedge meadow, wet prairie, and wet mesic prairie into the wet meadow category. I assigned units described as mesic prairie, grassland, savanna, or simply as buffer into the buffer category for performance analysis. In some cases, the plant community type for sampling units was not clearly identified or was reported as changing between buffer and wetland during the life of the bank. I chose to exclude these sampling units from my analysis.

I used the plant species lists and raw species frequency and cover data from vegetation quadrat monitoring to calculate the vegetation-based metrics necessary to determine if banks had met their performance standards. As stated previously, the numbers of quadrats and transects were not consistent between banks, so for each monitoring year at a single bank, and using data from only one monitoring visit, I combined transects to pool the quadrat data by plant community type. The ICA documents indicate that most standards are to be evaluated separately for each of the three plant community types, so I followed this method in my analysis rather than combining results across community types in a single bank. Not every bank contained each community type, so the sample size differed for each community. My analysis of performance in the final monitoring year (Objective 1) included a total of 21 bank sites, and the number of sites for each habitat type were as follows: 9 marsh sites, 20 wet meadow sites, and 13 buffer sites. I was able to include a total of 12 bank sites in my temporal analysis (Objective 2), which included 5 marsh sites, 12 wet meadow sites, and 5 buffer sites.

A summary of all the performance standards described in the 1997 and 2008 ICA documents is provided in Table 2.1. I determined mitigation bank performance relative to the standards measuring native perennial richness, invasive importance, invasive dominant species, native perennial importance, all natives importance, the mean Coefficient of Conservatism for native species (native mean C), and Floristic Quality Index for native species (native FQI). I did not analyze standards related to the establishment of jurisdictional wetlands, the frequency of native perennial species, vegetation cover, or wetland hydrology due to insufficient or inconsistent data in the bank monitoring reports. I applied the performance standard definitions and compliance thresholds given in the 1997 and 2008 ICA documents for each performance standard as follows:

Native Perennial Richness: For this standard, I totaled the number of native perennial species within each plant community type. Based on the language of the ICA documents and the approach taken by the bank sponsors this metric should be calculated using both the quadrat data and the timed meander search species list for each community type, but I was only able to use quadrat data in calculating this standard. The minimum number of native perennial species required by the IRT to meet this standard is different for each community type: 15 species in marshes, 35 species in wet meadows, and 25 species in buffers.

Invasive Importance: This standard establishes a limit on the dominance, measured by relative importance value, by invasive and non-native species that is acceptable in banks.

Following the approach described in the performance standards and used by the bank sponsors, I calculated relative importance value for each species as the average of relative cover and relative frequency for that species. This value, expressed here as a percentage, represents a measure of dominance for each plant species. To meet this standard, the cumulative relative importance value of all invasive and non-native species could be no greater than 5% for each plant community. Under the 1997 guidelines, the species restricted under this standard included all *Typha* taxa, *Phalaris arundinacea*, and all other non-native species. It is noteworthy that all three *Typha* taxa in the region (*Typha angustifolia*, *Typha* x *glauca*, and *Typha latifolia*) were restricted under this 1997 standard despite the fact that they were considered to be native at the time (Swink and Wilhelm 1994). *Typha angustifolia* and *Typha* x *glauca* have more recently been recognized as an exotic species and a native-exotic hybrid (Wilhelm and Rericha 2017). I also note that *Phragmites australis*, a relatively common grass that is now considered to have both a native and an abundant non-native subspecies in the Chicago region (Wilhelm and Rericha 2017), was classified solely as a native species during the operation of these banks, and so was not restricted under this performance standard (Swink and Wilhelm 1994). To duplicate the conditions under which the banks in my study were originally evaluated, I chose to treat all *Typha* taxa and *Phragmites australis* as native species for all standards, though I still restricted *Typha* taxa under the standards limiting invasive species.

Invasive Dominant Species: The 2008 ICA applied an additional standard that limited the acceptable dominance by non-native or weedy species. This standard does not allow for any of the three most dominant species (measured by relative importance value) in any plant community at a bank to be a non-native or restricted species (including *Phragmites australis, Typha angustifolia, Typha* x *glauca,* and the native *Salix interior*). I determined bank compliance with this standard by calculating the relative importance value for each species in a plant community and found the bank to have met the standard if the species with the three highest values were not among the restricted species.

Native Perennial Importance and All Natives Importance: This standard establishes a minimum value for the dominance, measured by relative importance value, of native species in banks. The performance standard required that native species cumulatively accounted for at least 80% of the relative importance value in each plant community. Under the 1997 ICA, this standard could only be met using dominance by perennial native species. In the monitoring reports for banks that were operated under the 1997 ICA, bank sponsors often suggested that this metric should be expanded to include annual and biennial native species as well. The 2008 ICA allowed for this adjustment if certain conditions were met. I tested both versions of this standard, including only native perennial species under the 1997 ICA conditions and including all native species under the conditions of the 2008 ICA.

Native Mean C and Native FQI: These standards, which were present only in the 2008 ICA, require that banks establish plant communities with a certain level of floristic quality, based on indices derived from coefficients of conservatism (C-values) (Swink and Wilhelm 1994). C-values had previously been assigned to each native plant species present in the Chicago region by botanists familiar with the regional flora. These values range from 0 to 10 and were designed to indicate a species' tolerance to anthropogenic disturbance and fidelity to high quality natural habitats. High scores were given to conservative species found exclusively in undegraded natural communities, whereas species given low scores readily inhabit sites that have experienced anthropogenic disturbance, though they may occur in undegraded natural communities as well. Performance standards used in the 2008 ICA are based on two metrics calculated from C-values:

native mean C and native Floristic Quality Index (FQI). Native mean C is the average of the C-values for all the native plant species found at a site. Native FQI is derived from native species richness as well as floristic quality, and is calculated as:

$$\text{native FQI} = \text{native mean C} \times \sqrt{S}$$

$$S = \text{the total number of native plant species at a site}$$

The 2008 ICA indicates that the floristic quality standards should be evaluated for each plant community type and for the entire bank site, but I only assessed these standards for individual plant communities. These standards would ideally be determined using species lists obtained from both quadrat data and timed meander searches, but I was only able to use quadrat-monitoring data for my calculations. To meet this standard, native mean C values were required to exceed 3.5 and native FQI values were to exceed 20 in each plant community type. The native perennial richness performance threshold is scaled by plant community type, indicating that the IRT expects species richness to vary by community; however, the native FQI performance threshold is the same for each community type, despite the fact that this metric is partially derived from species richness. This suggests that the native FQI standard may be easier to meet in community types that possess inherently greater richness of native species.

Numerical analysis

For my analysis of bank performance in the final year of monitoring, I calculated the absolute bank compliance for each plant community type by determining the percentage of banks that met or exceeded the performance threshold for each standard. I also assessed the distribution of bank scores for each performance metric and tested if the mean bank scores for each metric exceeded the performance threshold. I conducted this analysis using one-sample, one-tailed t-tests for every performance standard except that measuring invasive dominant species; this standard represents a count of invasive species, so I tested it using a one-sample, one-tailed Poisson rate test. My null hypotheses for all one-sample hypothesis tests were that mean bank scores exceeded performance thresholds (i.e. banks successfully met the performance standard) ($\alpha = 0.05$).

For my temporal analysis of change in vegetation metrics, I conducted paired, two-tailed t-tests ($\alpha = 0.05$) to determine if there was a significant difference in metric scores between selected monitoring years for every performance standard except for the invasive dominant species standard. I tested the invasive dominant species standard using a two-sample, two-tailed Poisson rate test ($\alpha = 0.05$).

2.3 Results

Performance in final monitoring year

No standard, in any habitat type, was met by every bank (Table 2.2). Bank performance was lowest for the standard limiting invasive and non-native species to 5% of the overall relative importance value, with fewer than 12% of banks meeting this standard for each plant community. For each plant community, the mean invasive relative importance value was at least 17% of the overall dominance and was, statistically, significantly higher ($P < 0.05$) than the 5% threshold for this standard (Figure 2.1B, Table 2.3). Comparing the two standards restricting invasive species, bank compliance was much higher for the invasive dominant species standard than for the invasive importance standard in wet meadow and buffer sites, though the dominant species standard remained very difficult for banks to meet in marsh habitats, with only one third of marsh sites meeting the performance threshold (Table 2.2). Poisson rate tests for the invasive dominant species standard showed that the mean number of invasive species among the three most dominant species at each site (0.78 in marsh sites, 0.45 in wet meadow sites, and 0.62 in buffer sites) was significantly greater than the permitted performance threshold of 0 for all community types ($P < 0.001$ in all cases).

Tables 2.4-2.6 list the species with the greatest relative importance for each plant community, showing which invasive and non-native species contributed the most to the failure of banks to meet the standards limiting invasive dominance. For each plant community, I found that the invasive species with the greatest relative importance was an annual exotic species (*Setaria faberi* in marsh, *Bromus japonicus* in wet meadow, and *Chenopodium album* in buffer). *Phalaris arundinacea* and all *Typha* taxa were specifically identified as restricted species in the text for these performance standards; the percentage of relative importance accounted for by these species was large in marsh sites (15.12% *Typha angustifolia*, 3.93% *Phalaris arundinacea*, and 3.15% *Typha latifolia*), and smaller but still significant in wet meadow sites (3.00% *Typha angustifolia*, 2.57% *Phalaris arundinacea*, and 1.29% *Typha latifolia*) and buffer sites (2.92% *Phalaris arundinacea* and 0.00% *Typha* taxa).

Compliance was generally highest for standards involving native species richness and dominance (Figure 2.1A, 2.1C, 2.1D, Table 2.2). The majority of marsh and wet meadow sites were able to achieve these standards, though they were met in fewer than half of buffer sites. The results from the two performance standards measuring native species dominance based on relative importance value show that the adjustment made in the 2008 ICA allowed for much higher bank compliance. When I followed the 1997 ICA and only accepted native perennial species for this standard, 30% to 50% of banks met the standard in all community types (Table 2.2). The percentage of banks meeting the standard jumped to 78% in marsh and 80% in wet meadow communities when I adopted the 2008 ICA policy of including annual and biennial native species in calculations for this standard, though banks' performance in buffer sites remained unchanged (Table 2.2).

Banks showed only a moderate capacity to meet standards relating to floristic quality; just over half of marsh and wet meadow sites and just under a quarter of buffer sites exceeded the performance threshold for native mean C (Figure 2.1E, Table 2.2). Bank compliance for native FQI was similar, except that wet meadow sites were much more compliant, meeting this standard 90% of the time (Figure 2.1F, Table 2.2). The greater average metric score and compliance rate for Native FQI in wet meadows was somewhat expected, as this plant community type had greater species richness than marsh and buffer sites.

I did not assess overall bank performance across all habitat types for individual banks due to the uneven distribution of habitat types in different banks; however, I observed that only one bank was able to achieve compliance with every standard in each habitat type in its final year of monitoring. This site was composed exclusively of wet meadow habitat, and final monitoring occurred in its third growing season following restoration.

Temporal change in performance metrics

The low sample sizes in the time series analysis, particularly for marsh and buffer communities (5 sites each), diminished the power of t-tests to test for change over time in vegetation metrics, but there is evidence of a temporal change for certain vegetation metrics. For all community types, there was an increase in the sample mean and median values between monitoring years for native perennial richness (Figure 2.2A), native mean C (Figure 2.2E), and native FQI (Figure 2.2F), except for the median values in buffer sites. This temporal change was statistically significant ($P < 0.05$) for wet meadow sites (Table 2.7). Clear trends between monitoring visits were not easily discernible for the remaining performance standards, except perhaps for a weak increase in native perennial importance in wet meadow sites (Figure 2.2C, Table 2.7). Poisson rate tests for the invasive dominant species standard did not show a significant difference between sample visits for marsh (P = 1.00), wet meadow (P = 1.00), or buffer sites (P = 0.23). The mean number of invasive species among the three most dominant species for the first and second monitoring visits, respectively, were 0.80 and 0.60 in marsh sites, 0.58 and 0.58 in wet meadow sites, and 1.60 and 0.60 in buffer sites.

2.4 Discussion

While a high percentage of banks met performance standards related to the number and importance of native species, standards addressing dominance by invasive species were met in very few banks, indicating that most banks were unable to achieve compliance with all their standards. Other studies that evaluated large numbers of permittee-responsible mitigation wetlands in Illinois have also found that meeting all performance standards is a difficult goal for wetland mitigation projects to reach (Matthews and Endress 2008, Van den Bosch and Matthews 2017); however, a study of wetland mitigation bank projects in Ohio did find that all vegetation performance standards were met in these sites, though only two bank sites were assessed (Spieles et al. 2006).

Influence of invasive species

It is clear from my study that wetland mitigation bank managers in Chicago struggled to limit the presence and abundance of invasive species to levels that were acceptable to the IRT. Banks failed to meet the performance standard limiting the relative importance value of invasive and non-native species far more than any of the other standards I assessed, as it was failed in nearly 89% of marsh sites, 90% of wet meadow sites, and 100% of buffer sites. Other studies have shown that, like wetland mitigation banks, permittee-responsible mitigation wetlands also frequently fail to meet performance standards related to non-native dominance (Matthews and Endress 2008, Van den Bosch and Matthews 2017). Studies of the ecological outcomes of wetland restoration have shown that dominance by non-native species in restored wetlands is often higher than that in natural wetlands (Brooks et al. 2005, Aronson and Galatowitsch 2008, Matthews and Spyreas 2010). Pressure on wetlands from non-native plants is high in the interior plains region of the United States, including Illinois, and is particularly high on herbaceous wetlands such as those included in my study (EPA 2016). Included in the mitigation bank monitoring reports I reviewed were summaries of the management activities performed in banks, and these showed that bank sponsors generally made consistent efforts to manage and remove non-native species from banks; however, my data show that these efforts were typically inadequate to bring banks into compliance with the performance threshold required for non-native dominance.

Annual exotic species such as *Setaria faberi*, *Setaria glauca*, *Bromus japonicus*, and *Chenopodium album* were, in all habitat types, the species that contributed most significantly to banks' failure to meet the invasive importance standard. All of these are weeds of waste ground and cultivated areas (Swink and Wilhelm 1994). Successional weeds such as these are characteristic of the vegetation communities in restored wetlands in the early years following restoration, but species composition generally shifts towards greater dominance by perennial species within the first few years (Matthews and Endress 2010) to the first two decades (Aronson and Galatowitsch 2008) following restoration. This pattern of succession from ruderal, annual species towards greater perennial dominance has previously been observed specifically in wetland mitigation banks (Stefanik and Mitsch 2012). This succession was likely proceeding in the mitigation banks I assessed, but the effect that annual weeds had in causing banks to fail the invasive importance standard indicates that the vegetation communities in bank sites had not yet reached a condition of perennial dominance to the degree required by regulators at the time of final monitoring. This shortfall may be due especially to banks that completed final monitoring and received final credit release as early as the end of their second growing season since restoration. A longer monitoring period would allow regulators to assess banks more effectively after these successional changes have progressed further (Stefanik and Mitsch 2012). While the loss of exotic annual weeds in favor of perennial species is expected to continue in restored wetlands in banks, it is not clear that species composition in restored wetlands will reach a point of equivalence with natural wetlands, as annual species have continued to distinguish some wetland restorations from natural reference wetlands (Matthews and Spyreas 2010, Stefanik and Mitsch 2012) even as long as 19 years following restoration (Aronson and Galatowitsch 2008).

The IRT specifically identified *Phalaris arundinacea* and *Typha* taxa as undesirable species in the performance standards limiting invasive species that were used to evaluate the banks in my study (IRT 1997). While not the most dominant invasive or non-native species in the banks I assessed, *Phalaris arundinacea* and *Typha* spp. were abundant enough in all habitat types, especially marshes, to contribute significantly to banks' inability to meet the 5% threshold for invasive importance. Other studies have identified *Phalaris arundinacea* and *Typha angustifolia* as the species most responsible for very low compliance with exotic species standards in mitigation wetlands (Matthews and Endress 2008, Van den Bosch and Matthews 2017). The influence of these species on the banks in my study was somewhat less than has been observed by others in mitigation, restored, and natural wetlands; however, there is much evidence that *Phalaris arundinacea* in particular can become increasingly dominant in wetlands. *Phalaris arundinacea* has been found to increase over time, become a dominant species, and cause regional homogenization of wetlands in the Midwest (Aronson and Galatowitsch 2008, Matthews and Spyreas 2010, Price et al. 2017). In mitigation wetlands, compliance with standards may decrease over time following the end of typical monitoring periods due to *Phalaris arundinacea* dominance (Van den Bosch and Matthews 2017*). Phalaris arundinacea* dominance is associated with undesirable conditions in vegetation communities, including lower species richness and floristic quality (Spyreas et al. 2010). I noted that banks sponsors made consistent efforts to remove *Phalaris arundinacea* and undesirable *Typha* taxa from banks. This may explain the relatively low abundance of these species in banks I evaluated, relative to the values presented in some of the studies cited above; however, it is possible that some of the very young banks included in my study simply had not had sufficient time pass to become thoroughly invaded. While the 5% threshold for relative importance of invasive species represented a very demanding standard, and was unmet by most banks, it does seem clear that banks must be required to demonstrate extensive control of highly aggressive species such as *Phalaris arundinacea* in order to ensure that the wetlands in banks avoid the progression towards *Phalaris arundinacea* dominance that is common to many wetlands in the region.

Standards measuring native species

The banks in my study showed high rates of compliance (greater than 75% in marsh and wet meadow communities) with standards measuring native perennial richness and relative importance of all native species, while compliance for relative importance of native perennial species and floristic quality standards was closer to 50% in wetland communities. These results match closely those found by Matthews and Endress (2008) in permittee-responsible mitigation wetlands in Illinois, though Van den Bosch and Matthews (2017) found somewhat higher compliance with floristic quality standards in a similar study. It is worth noting that the threshold for relative importance value of native species required in banks (80%) is much higher than what was required of permittee-responsible mitigation wetlands in these other studies (50%). Some of the performance standards I assessed were based solely on vegetation quadrat data, but native perennial richness, native mean C, and native FQI were designed to incorporate species lists collected from both the quadrats and the wider timed meander search of each sampling unit. Since I was not able to include these timed meander search data, my assessment of performance was based on a smaller sampling area (quadrats only) than was intended (the entire sampling unit). This likely caused me to underestimate species richness in sampling units. I reported a high level of compliance with the native perennial richness standard in emergent and wet meadow sites, but it is reasonable to expect that compliance for this standard in these wetland community types may have been close to 100% if I had been able to include the full species lists in my analysis. Native FQI is partially derived from native species richness, so the values I report for native FQI may also be lower than those that would have been reported if I had included timed meander search data. Measurements of mean C, however, are relatively unaffected by changes in sampling area (Spyreas 2016), so my compliance estimates for this standard were likely not compromised by using only quadrat-level data.

Temporal change in bank vegetation

I found some evidence that native perennial species richness, native mean C, and native FQI increased over time during mitigation banks' required monitoring period. Assessing the trajectories of vegetation-based metrics such as these in restored wetlands can be difficult due to complex temporal trends. Species richness in wetland mitigation banks has been shown to fluctuate during the first five years following restoration (Spieles 2005, Spieles et al. 2006), but in studies of some restored wetlands, significant changes in species richness, both positive and negative, continued for longer than 10 years after project construction (Mulhouse and Galatowitsch 2003, Aronson and Galatowitsch 2008, Gutrich et al. 2009). Even vegetation surveys of mitigation wetlands in consecutive years may yield significant changes in the number and composition of species observed (Wall and Stevens 2015). Variable trajectories have been observed in mitigation wetlands for floristic quality metrics as well, as different studies have observed significant increases (Matthews and Endress 2008), decreases (Van den Bosch and Matthews 2017) and variability (Spieles et al. 2006) over time, both during and after the typical five year monitoring period. Due to the temporal shifts in these metrics, especially in young mitigation wetlands, many have recommended that the typical five-year monitoring period for mitigation wetlands should be extended or restructured (Spieles et al. 2006, Matthews et al. 2009, Stefanik and Mitsch 2012, Morgan and Hough 2015, Van den Bosch and Matthews 2017). Temporal changes for certain metrics in my study, and especially the strong influence I observed of annual non-native species on banks, indicate that community succession and assembly was still occurring at the time of final monitoring in the banks I assessed. As such, a longer monitoring period would have been beneficial, especially for sites that were evaluated sooner than the typical five-year monitoring period.

Buffer communities

Comparing bank performance between different plant community types was not one of my primary objectives, but it is noteworthy that marsh and wet meadow communities in banks showed much higher performance metric scores and percent compliance than buffer communities for nearly all performance standards. The goal of wetland mitigation banking is to produce wetland resources, so it is likely that both bank sponsors and regulatory agencies prioritize the restoration of marsh and wet meadow communities over that of buffers. The majority of the area in wetland mitigation banks in the Chicago District consists of wetland communities rather than buffers, and regulatory policies limit the amount of credits that may be generated by buffer areas to 15% of the total credits produced by the bank (IRT 2008, IRT 2017). The maximum credits per acre that the IRT grants for buffer communities is also lower than that of restored wetland communities. As a result, bank sponsors may devote fewer management resources to buffers than to wetland areas, which could explain the poor performance of buffers relative to marsh and wet meadow communities. It is also possible that the regulatory agencies may not hold buffer communities to the same performance expectations that they demand of marsh and wet meadow communities, if their priority is to ensure that banks are producing wetlands of acceptable quality.

Bank evaluation by regulatory agencies

The performance standards used by regulatory agencies have been adapted over time. In 2017, the Chicago District issued an updated set of performance standards (IRT 2017) that is much more complex and specific than previous versions. Regulatory agencies' work of adapting and improving performance standards over time has likely been shaped by their observations of past performance in banks, and by feedback from bank sponsors. My results suggest that these feedback mechanisms may have influenced changes in Chicago District performance standards. I found that very few banks were able to meet the invasive importance standard established under the 1997 ICA. The Chicago District performance standards issued in 2017 contain a similar standard, but the allowable threshold for relative cover of invasive and non-native species has increased from 5% in 1997 to 10% in 2017. This change suggests a willingness by the IRT to adjust standards to levels that are more realistically achievable for wetland mitigation banks. The performance standard requiring that banks meet a minimum threshold for native species dominance has also changed over time. In 1997 the IRT required that only native perennial species could be used to meet this standard. In my review of bank monitoring reports, I noted that bank sponsors often suggested that annual and biennial native species constituted desirable wetland resources and should be allowed to count for this standard. These comments likely influenced Chicago District policies, as the 2008 version of this standard included a provision that allowed native annual and biennial species to be used to meet this standard (IRT 2008). My results show that this change resulted in higher levels of bank compliance with this standard.

I recognize that inconsistent data collection protocols among banks likely introduces some variation in the performance data for which I was unable to account. Variability in the structure and reporting of mitigation documents adds complexity to the task of evaluating wetland mitigation (Fennessy et al. 2013). The lack of consistency in vegetation monitoring and reporting at wetland mitigation banks has caused difficulty for regulators attempting to evaluate bank compliance with permit conditions (Reiss et al. 2009) and been a limiting factor in other assessments of banks (Spieles 2005).

Only one bank in my study met all its performance standards in every habitat type; however, every site I assessed received final approval and credit release from the IRT, even though compliance with all performance standards is stated in regulatory documents as a requirement for final release of credits (Corps and EPA 2008). When banks did not meet all their performance standards, the IRT would sometimes require that the bank sponsor continue to manage and monitor the site until it was able to meet its standards. In some cases, however, the IRT would grant a final release of credits for banks that had not met all standards, but would decrease the total number of credits released from the bank in order to account for areas of the site that were not compliant. This practice apparently allows the IRT to make credit release decisions based on its own observations of the condition and trajectory of wetlands in mitigation banks, rather than making these decisions based exclusively on bank compliance with numerical performance standards.

Conclusion

A broad national review of wetland mitigation through the end of the 20th century by the National Research Council (NRC 2001) found that mitigation projects did not meet performance standards, and in some cases were not even constructed, frequently enough that mitigation policies were producing a net loss of wetland resources. More recent work (Matthews and Endress 2008) has also shown that permittee-responsible mitigation projects did not produce their required wetland area, often because of a failure to generate sufficient wetland hydrology. The Corp and EPA's mitigation rule (2008), issued partly in response to the National Resource Council's findings, established a preference for wetland mitigation banking. One benefit of mitigation banking that was provided as a rationale for this change was that banks primarily generate wetland resources in advance of the destruction of natural wetlands for which they will be used as compensation. This is assumed to increase the likelihood that adequate compensation will be provided for wetland losses, compared to permittee-responsible mitigation that occurs after the destruction of natural wetlands has already occurred. While I did not consider the amount of wetland area produced by banks in this study, my results and my observations of wetland mitigation practice do suggest that banking policy in the Chicago District is applied in a way that safeguards against some of the past failings of wetland mitigation that led to a net loss in wetland area. I found that most banks in my study failed to meet all their performance standards, but that this lack of compliance did result in reductions to the final number of credits released to banks, rather than allowing noncompliant wetland mitigation to be used as compensation for wetland impacts.

However, this conclusion is complicated by credit releases from banks that occur prior to final monitoring. In an examination of wetland mitigation banks in Florida, Reiss et al. (2009) found that banks' compliance with permit criteria was quite high, but that this compliance reflected project completion more than ecological outcomes. For the banks in their study, 60-75% of credit releases were tied to the completion of legal, construction, and management activities at the banks, while only 25-40% of credit releases came after the bank demonstrated hydrologic or ecological responses that met permit criteria. Their findings are similar to the credit release schedule formerly used to regulate banks in the Chicago District. Under the 1997 ICA, privately-sponsored banks received up to 30% of their credits after the bank was approved and appropriate financial assurances were provided, an additional 20% of their credits once appropriate wetland hydrology was demonstrated following project construction, an additional 20% of their credits once the project's approved planting and seeding plan had been completed, and the final 30% of credits after the bank was found to have met it ecological performance standards (IRT 1997). Under these regulatory conditions, the release of credits to bank sponsors is tied to the ecological outcomes of wetlands in banks for fewer than half of the total credits released to banks. However, the Chicago District IRT has more recently increased the percentage of bank credits that require the demonstration of appropriate ecological outcomes for their release. Under the 2017 ICA, in privately-sponsored banks, a maximum of 20% of credits may be released following initial project approval and site protection, 25% of credits may be released once wetland hydrology at the site has been demonstrated, and the remaining credits are released only after the bank has met interim and final performance standards based primarily on vegetation (IRT 2017).

This study has provided a quantitative assessment of wetland mitigation banks' compliance with performance standards at or near the end of their monitoring periods and an examination of banks' ability to meet individual performance standards. Regulatory agencies have shown an ability and willingness to adapt wetland mitigation policies over time based on the ecological and compliance outcomes of previous policy conditions. The results of this study can serve as a useful resource for the continued adaptation and improvement of performance standards and mitigation assessment.

2.5 Figures

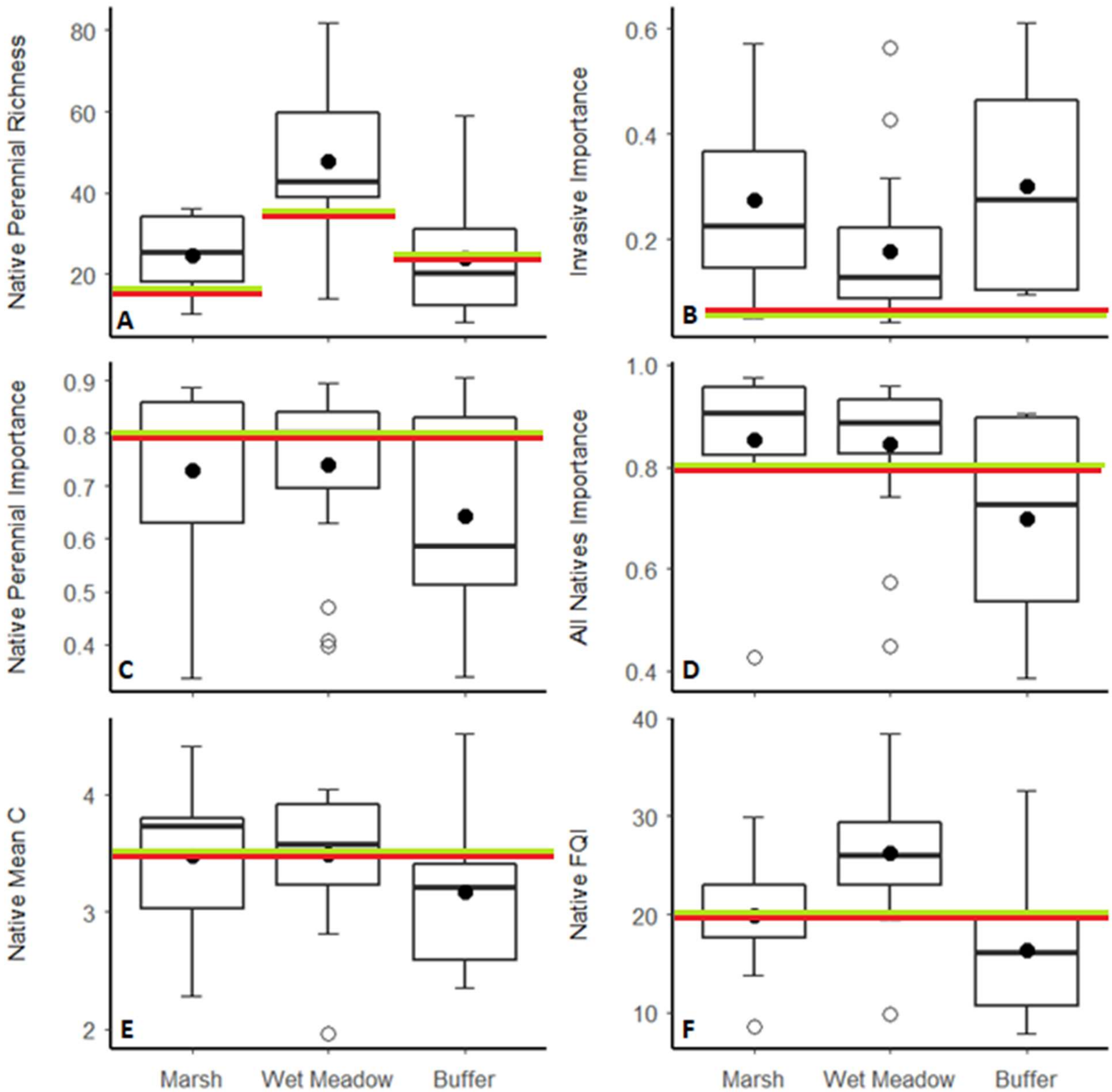

Figure 2.1. Box-and-whisker plots illustrating performance standard metrics in wetland mitigation banks for each regulatory habitat type. Results were calculated from the final quadrat monitoring data collected from banks. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). Colored lines show the performance threshold, with the green side indicating the area of compliance and the red side indicating the area of non-compliance. The number of banks included in this analysis for each habitat type are: 9 marsh sites, 20 wet meadow sites, and 13 buffer sites.

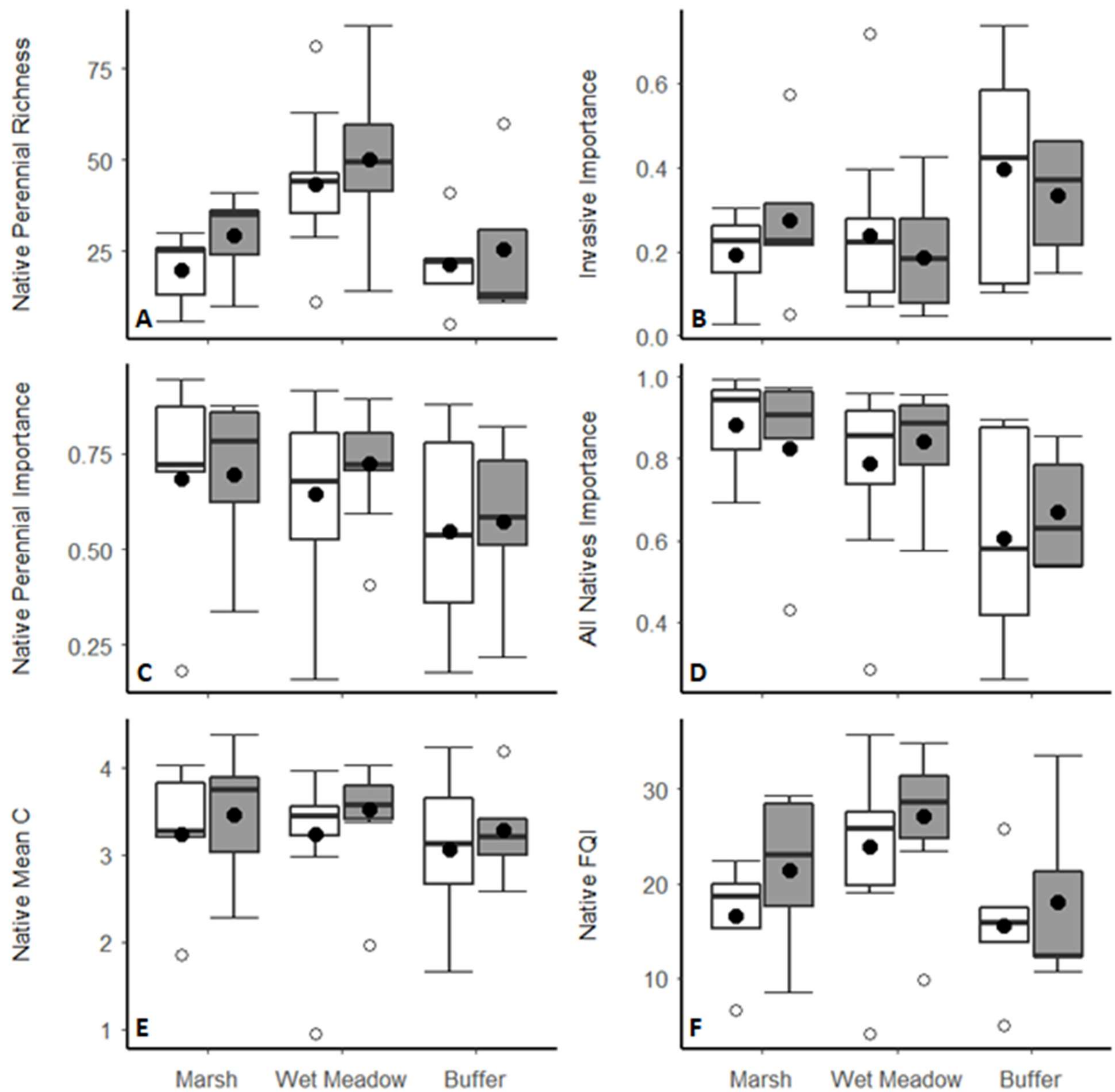

Figure 2.2. Box-and-whisker plots illustrating the change in performance standard metrics between monitoring visits at wetland mitigation banks, for each habitat type. Banks included in this analysis had paired monitoring visits, with the first visit occurring between the bank's second and fourth growing season, the second visit occurring between the fourth and sixth growing season, and with either a two- or three-year gap between monitoring visits. Data from the first monitoring visit are shown by unshaded boxplots and data from the second monitoring visit are shown by shaded boxplots. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). The number of banks included in this analysis for each habitat type are: 5 marsh sites, 12 wet meadow sites, and 5 buffer sites.

2.6 Tables

Table 2.1. Performance standards used in the 1997 and 2008 ICA by the Chicago District of the Corps. Standards assessed in this study are marked with an asterisk.

Category	1997 ICA	2008 ICA
Jurisdictional Wetlands	Wetlands created/restored for credit must meet the criteria for jurisdictional wetlands	Same as 1997 ICA
*Native Perennial Richness	A minimum number of native perennial species must be present within each plant community: -Marsh: minimum of 15 species -Sedge meadow/wet prairie: minimum of 35 species -Mesic prairie (buffer): minimum of 25 species	Same as 1997 ICA
Species Frequency	At least 50% of the required minimum number of species must be present at 10% frequency or greater for each plant community	Same as 1997 ICA
*Invasive Importance	Cumulative relative importance value of *Typha* spp., *Phalaris arundinacea*, and non-native species shall be less than 5% of the total dominance for each plant community	Cumulative total percent cover (not relative cover) of non-native or weedy species[a] shall not be more than 5% for each plant community

a Includes but is not limited to *Typha angustifolia*, *Typha* x *glauca*, *Phragmites australis*, *Lythrum salicaria*, *Salix interior*, and *Phalaris arundinacea*

Table 2.1. (cont.)

Category	1997 ICA	2008 ICA
*Invasive Dominant Species	None	None of the three most dominant plant species in any wetland plant community may be non-native or weedy species[a]
*Native Perennial Importance and *All Natives Importance	Native perennial species within each wetland plant community shall have a relative importance value of at least 80%	Same as 1997 ICA. However, a lower percent native perennial dominance measure may be acceptable if it is demonstrated that the remaining dominance percentage is by native annual and biennial wetland plant species
*Native Mean C	None	A native mean C value of greater than or equal to 3.5 must be achieved in each separate plant community and as measured over the entire mitigation bank site
*Native FQI	None	The native FQI value must be greater than or equal to 20 in each separate plant community and as measured over the entire mitigation bank site

a Includes but is not limited to *Typha angustifolia*, *Typha* x *glauca*, *Phragmites australis*, *Lythrum salicaria*, *Salix interior*, and *Phalaris arundinacea*

Table 2.1. (cont.)

Category	1997 ICA	2008 ICA
Vegetation Cover	None	No area over the entire mitigation bank site greater than 1 square meter shall be devoid of vegetation, as measured by aerial coverage, unless specified on approved mitigation plans. This standard does not apply to emergent and aquatic communities
Wetland Hydrology	Wetland hydrology must be independently demonstrated within each wetland from data gathered from piezometers placed throughout the bank site	All wetland plant communities receiving credit shall have soils saturated within 12 inches or less of the ground surface for at least 12.5% of the growing season. To meet this standard, the bank must demonstrate inundated or saturated soil for 23 consecutive days during the growing season.

Table 2.2. Percentage of banks that met the 1997 and 2008 ICA Performance Standards. Performance was calculated using the final quadrat monitoring data from each bank. Performance was calculated separately for each of the three regulatory plant community types: Marsh (9 sites), Wet Meadow (20 sites), and Buffer (13 sites).

Performance Standard	ICA Version	Plant Community		
		Marsh	Wet Meadow	Buffer
Native Perennial Richness	1997, 2008	77.78%	85.00%	46.15%
Invasive Importance	1997	11.11%	10.00%	0.00%
Invasive Dominant Species	2008	33.33%	70.00%	69.23%
Native Perennial Importance	1997, 2008	44.44%	50.00%	30.77%
All Natives Importance	2008	77.78%	80.00%	30.77%
Native Mean C	2008	55.56%	55.00%	23.08%
Native FQI	2008	44.44%	90.00%	23.08%

Table 2.3. Parameter estimates and test statistics obtained from one sample one-tailed t-tests comparing bank performance in the final monitoring year to the threshold for each performance standard in each plant community type. The null hypothesis tested for each standard was that the banks successfully met the standard. The number of bank sites included for each plant community type were: 9 Marsh, 20 Wet Meadow, and 13 Buffer.

Plant Community	Performance Standard	Threshold	Mean	*T*	*P*
Marsh	Native Perennial Richness	> 15	24.56	3.02	0.9917
	Invasive Importance	< 5%	27.37%	3.94	0.0022
	Native Perennial Importance	> 80%	73.07%	-1.16	0.1401
	All Natives Importance	> 80%	85.27%	0.92	0.8081
	Native Mean C	> 3.5	3.48	-0.11	0.4572
	Native FQI	> 20	19.83	-0.08	0.4685
Wet Meadow	Native Perennial Richness	> 35	47.65	3.30	0.9981
	Invasive Importance	< 5%	17.54%	4.16	0.0003
	Native Perennial Importance	> 80%	74.01%	-1.74	0.0487
	All Natives Importance	> 80%	84.55%	1.55	0.9309
	Native Mean C	> 3.5	3.49	-0.06	0.4753
	Native FQI	> 20	26.23	4.37	0.9998
Buffer	Native Perennial Richness	> 25	23.92	-0.27	0.3959
	Invasive Importance	< 5%	30.11%	4.89	0.0002
	Native Perennial Importance	> 80%	64.40%	-2.82	0.0078
	All Natives Importance	> 80%	69.89%	-1.97	0.0364
	Native Mean C	> 3.5	3.17	-1.72	0.0555
	Native FQI	> 20	16.27	-2.01	0.0336

Table 2.4. The 25 plants species with the highest relative importance value, averaged across all marsh sites, calculated using the final quadrat monitoring data from each bank. We calculated relative importance value for each bank site as the average of relative cover and relative frequency for each plant species. Native C-values are from Swink and Wilhelm (1994).

Species	Native C-Value	Relative Importance
Setaria faberi[a]	-	16.56%
Typha angustifolia[b]	1	15.12%
Leersia oryzoides	4	9.96%
Asclepias syriaca	0	9.46%
Sagittaria latifolia	4	9.26%
Andropogon gerardii	5	8.69%
Ambrosia artemisiifolia var. *elatior*	0	6.72%
Alisma subcordatum	4	6.58%
Eleocharis erythropoda	2	6.04%
Sparganium eurycarpum	6	5.84%
Agropyron repens[a]	-	5.34%
Eleocharis acicularis	2	5.30%
Convolvulus arvensis[a]	-	5.05%
Ranunculus sceleratus	6	4.85%
Lemna minor	5	4.54%
Scirpus fluviatilis	4	4.53%
Juncus dudleyi	4	4.47%
Solidago canadensis	1	4.38%
Phalaris arundinacea[a]	-	3.93%
Carex vulpinoidea	2	3.88%
Echinochloa crusgalli	0	3.84%
Juncus torreyi	4	3.79%
Bromus japonicus[a]	-	3.75%
Boltonia latisquama var. *recognita*	9	3.58%
Carex scoparia	7	3.26%

a Exotic species

b Considered native, but restricted by the performance standard limiting exotic and weedy species

Table 2.5. The 25 plants species with the highest relative importance value, averaged across all wet meadow sites, calculated using the final quadrat monitoring data from each bank. We calculated relative importance value for each bank site as the average of relative cover and relative frequency for each plant species. Native C-values are from Swink and Wilhelm (1994).

Species	Native C-Value	Relative Importance
Bidens coronata	9	9.21%
Carex tribuloides	3	7.76%
Elymus virginicus	4	5.26%
Lysimachia ciliata	4	4.55%
Aster sp.	-	4.40%
Solidago canadensis	1	4.35%
Carex granularis	4	4.05%
Carex cristatella	4	3.78%
Aster simplex	3	3.62%
Bromus japonicus[a]	-	3.49%
Silphium perfoliatum	5	3.47%
Leersia oryzoides	4	3.32%
Scirpus fluviatilis	4	3.32%
Andropogon gerardii	5	3.30%
Carex aquatilis var. *altior*	5	3.29%
Solidago altissima	1	3.22%
Panicum virgatum	5	3.08%
Boltonia latisquama var. *recognita*	9	3.06%
Spartina pectinata	4	3.03%
Typha angustifolia[b]	1	3.00%
Bidens cernua	5	3.00%
Scirpus pungens	5	2.95%
Phalaris arundinacea[a]	-	2.57%
Carex comosa	5	2.57%
Eleocharis erythropoda	2	2.42%

a Exotic species

b Considered native, but restricted by the performance standard limiting exotic and weedy species

Table 2.6. The 25 plants species with the highest relative importance value, averaged across all buffer sites, calculated using the final quadrat monitoring data from each bank. We calculated relative importance value for each bank site as the average of relative cover and relative frequency for each plant species. Native C-values are from Swink and Wilhelm (1994).

Species	Native C-Value	Relative Importance
Andropogon gerardii	5	17.51%
Chenopodium album[a]	-	8.48%
Carex cristatella	4	7.57%
Solidago altissima	1	7.49%
Agropyron repens[a]	-	7.32%
Sorghastrum nutans	5	6.42%
Panicum virgatum	5	6.13%
Leersia oryzoides	4	6.07%
Solidago canadensis	1	5.63%
Rudbeckia triloba	3	5.51%
Bromus japonicus[a]	-	5.36%
Setaria glauca[a]	-	5.12%
Monarda fistulosa	4	4.94%
Ratibida pinnata	4	4.61%
Poa pratensis[a]	-	4.55%
Fragaria virginiana	1	4.16%
Cirsium arvense[a]	-	4.08%
Aster simplex	3	4.03%
Trifolium hybridum[a]	-	3.77%
Echinacea purpurea	3	3.63%
Carex tribuloides	3	3.59%
Aster pilosus	0	3.51%
Carex bebbii	6	3.24%
Rudbeckia hirta	1	3.19%
Elymus canadensis	4	3.03%

a Exotic species

Table 2.7. Sample means and test statistics obtained from paired two-tailed t-tests comparing vegetation metrics in banks between two monitoring visits in each plant community. The null hypothesis for each test was that there was no difference in the vegetation metric means between paired monitoring visits. The number of bank sites included for each plant community type were: 5 Marsh, 12 Wet Meadow, and 5 Buffer.

Plant Community	Performance Standard	First Mean	Second Mean	*T*	*P*
Marsh	Native Perennial Richness	20.00	29.20	-1.81	0.1446
	Invasive Importance	19.36%	27.52%	-1.59	0.1865
	Native Perennial Importance	68.40%	69.43%	-0.12	0.9139
	All Natives Importance	88.45%	82.39%	1.03	0.3628
	Native Mean C	3.23	3.46	-1.85	0.1385
	Native FQI	16.59	21.38	-1.87	0.1349
Wet Meadow	Native Perennial Richness	43.50	50.17	-2.71	0.0203
	Invasive Importance	23.77%	18.55%	1.60	0.1374
	Native Perennial Importance	64.32%	72.53%	-1.95	0.0775
	All Natives Importance	78.91%	84.30%	-1.66	0.1252
	Native Mean C	3.23	3.51	-2.28	0.0433
	Native FQI	23.96	27.06	-2.70	0.0205
Buffer	Native Perennial Richness	21.40	25.40	-0.77	0.4866
	Invasive Importance	39.46%	33.22%	0.94	0.3986
	Native Perennial Importance	54.61%	57.31%	-0.31	0.7724
	All Natives Importance	60.54%	66.89%	-0.97	0.3888
	Native Mean C	3.06	3.28	-0.64	0.5562
	Native FQI	15.59	18.05	-1.01	0.3714

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CHAPTER 3: COMPARING THE PLANT COMMUNITIES IN WETLAND MITIGATION BANKS TO THOSE IN NATURAL WETLANDS

3.1 Introduction

In the United States, the federal government regulates impacts to wetlands through wetland mitigation policy that is originally based upon Section 404 of the Clean Water Act of 1972. This policy is administered primarily by the US Army Corps of Engineers (Corps) and the US Environmental Protection Agency (EPA). All development and construction projects that cause adverse effects to wetlands must be permitted by the Corps. Permit applicants must compensate for unavoidable wetland impacts by creating or funding a project that generates wetland resources and values, an action referred to as compensatory mitigation. Compensatory mitigation is used by the Corps to support the specific goal of achieving “no overall net loss of values and functions” of wetlands (Corps and EPA 1990).

There are three mechanisms by which permittees may provide compensatory wetland mitigation: permittee-responsible mitigation, in-lieu fee mitigation, and mitigation banking (IWR 2015). When using permittee-responsible mitigation, permittees themselves are responsible for constructing a project that satisfies their mitigation requirements, though they may hire a restoration firm to complete the project. Under in-lieu fee mitigation, multiple permittees provide compensation by paying an in-lieu fee sponsor, which must be a government agency or non-profit conservation organization, so that the sponsor may construct a wetland mitigation project using the pooled funds. Wetland mitigation banks are large wetland compensation projects that are usually constructed by private, third-party sponsors. Banks sponsors construct banks in order to generate wetland credits, which can be sold at a profit to permittees to meet their mitigation requirements. In 2008, the Corps identified mitigation banking as its preferred mechanism for compensating for wetland impacts (Corps and EPA 2008), which has led to an increase in the use of banks. From 2010 to 2014, new permittees provided compensation for their wetland impacts by using mitigation bank credits 41% of the time, but the use of banks increased throughout this period, and in 2014 permittees used mitigation banking more than any other mitigation mechanism (IWR 2015). The percentage of overall wetland mitigation using mitigation banks has continued to increase from 2014 to 2017 (Hough and Harrington 2019).

Credits may be generated in compensatory wetland mitigation banks by four methods: restoration of wetland function to a non-wetland site that formerly existed as wetland; creation of wetlands in an upland site; enhancement of wetland function in an existing, degraded wetland; and preservation activities that remove threats to an existing wetland (IWR 2015). The Corps has determined restoration to be the preferred method of compensation because it generates new wetland area and may provide the greatest likelihood of success (Corps and EPA 2008).

Wetland mitigation banking is regulated by the Corps using standards and procedures that were formalized by a 2008 Final Compensatory Mitigation Rule (Corps and EPA 2008). The Corps determines the number of credits that are generated by banks and when they may be released. To receive final credit release, a bank must meet a set of ecological performance standards that are determined by the Corps (Corps and EPA 2008). The most common performance standards are vegetation-based metrics, though standards related to hydrology and the presence of jurisdictional wetlands are used as well (Environmental Law Institute 2002, Matthews and Endress 2008, Robertson and Hayden 2008, Reiss et al. 2009). The Corps evaluates bank compliance with performance standards using ecological monitoring data collected by the bank sponsor (Corps and EPA 2008). The sponsor must manage and monitor the bank for a minimum of five years, after which the Corps may release all remaining credits if the site has met its performance standards. Once credits have been released, the bank sponsor must transfer the property to an approved long-term owner and manager (e.g. a public land management agency). The sponsor must also provide a long-term management plan, financial mechanisms to fund long-term management activities, and a mechanism such as a conservation easement to ensure that the site has long-term legal protection (Corps and EPA 2008, Thomas 2016).

To define goals for all types of mitigation wetlands, and to judge if these wetlands are meeting their goals, it must be determined how mitigation wetlands should be evaluated. An answer that has often been given, both by regulatory agencies and by independent researchers, is that mitigation projects should be assessed in comparison to natural reference sites of the same wetland type (Brinson and Rheinhardt 1996, Brooks et al. 2005, Reiss et al. 2007, Fennessy et al. 2013). Comparing mitigation projects to a group of reference wetlands, rather than single sites, may be desirable so that the natural variability present among reference wetlands is considered (Kentula et al. 1992). Different approaches have been used to determine if reference wetlands should include degraded natural wetlands. National Corps regulations indicate that using reference wetlands to determine performance standards is beneficial, and that reference wetlands should reflect the existing range of natural and anthropogenic disturbance so that performance standards are appropriately achievable (Corps and EPA 2008). In comparison, certain regional agreements state that mitigation banks should create wetlands that are comparable to natural wetlands of moderate to high quality (IRT 2017). Some independent studies have supported the use of high-quality reference wetlands, which have been subject to minimal degradation, as a way to evaluate mitigation projects (Brinson and Rheinhardt 1996, Reiss et al. 2007). Alternatively, others have compared mitigation wetlands to reference sites that represented existing levels of degradation in natural wetlands (Campbell et al. 2002, Brooks et al. 2005, Matthews et al. 2009). When possible, it may be helpful to evaluate mitigation wetlands relative both to typical wetlands that include the full range of human disturbance present in a landscape and to high-quality reference wetlands that represent the least disturbed conditions (Fennessy et al. 2013).

Many independent studies have sought to determine if efforts to restore wetlands, both for mitigation projects and for other purposes, have produced plant communities that are equivalent with those in natural wetlands. The results have been variable, depending on the metrics used to evaluate vegetation and the age of restored wetlands. There may be temporal changes in the condition of restored wetlands relative to natural wetlands for several vegetation-based measures, including species richness (Gutrich et al. 2009, Hopple and Craft 2013), ecological integrity (Matthews et al. 2009, Van den Bosch and Matthews 2017) and species composition (Aronson and Galatowitsch 2008, Matthews and Spyreas 2010). While studies of compensatory wetland mitigation have usually focused on permittee-responsible mitigation, rather than mitigation banks (Morgan and Hough 2015), several studies have specifically examined the ecological condition of plant communities in banks. This work has sometimes shown that banks produce plant communities that have poor biological integrity relative to natural reference wetlands (Mack and Micacchion 2006), but there is also evidence that the ecological condition of banks may improve over time, eventually becoming similar to reference wetlands (Spieles et al. 2006, PGE and MBI 2012), at least in some bank sites (Stefanik and Mitsch 2012). The generality of these results may be limited for studies that primarily sampled banks that were less than 5 years old (Levrel et al. 2017), or that were only able to sample 2-5 banks.

Evaluations of the plant communities in wetland mitigation banks have been insufficient to determine if banks have produced wetlands that reach ecological equivalency with natural wetlands, especially for banks that have passed their typical five-year management and monitoring period (Levrel et al. 2017). Regulatory agencies often make their final evaluation of banks after five years, but ecosystem development and vegetation community assembly in restored wetlands cannot be sufficiently evaluated in this time frame (Zedler and Callaway 1999, Spieles 2005, Matthews et al. 2009, Stefanik and Mitsch 2012). In this study, I seek to assess the ecological outcomes of wetland mitigation banks by comparing the plant communities in banks that have passed their initial management and monitoring period to those in several categories of natural wetlands, representing both typical and high quality. The specific objectives of this study are to: (1) compare vegetation-based indicators of ecological integrity in mitigation bank wetlands to those in natural wetlands of variable quality, (2) determine if vegetation-based indicators of ecological integrity in mitigation bank wetlands are related to bank age, and (3) compare the plant community composition in mitigation bank wetlands to that in natural wetlands.

3.2 Methods

Wetland mitigation bank sampling

I collected vegetation data in the field from every wetland mitigation bank within the Chicago District of the Corps, which includes Cook, DuPage, Kane, Lake, McHenry, and Will counties in northeastern Illinois (Figure 3.1) that had received final credit certification by the end of August 2017, for a total of twenty banks. These banks were permitted and regulated by the Interagency Review Team (IRT), a group established by national mitigation regulations that includes representatives from local branches of the Corps, the EPA, and the US Fish and

Wildlife Service (IRT 2017). These banks were constructed between 1994 and 2009, and the time between final sign-off by the IRT and our field assessments ranged from less than one year to more than twelve years. Bank sites were between 29 and 97 acres in size. Some banks were constructed in separate, adjacent phases that were treated by the IRT as individual projects. In these cases, I randomly selected only one phase for sampling from each bank because adjacent phases could not be treated as independent sites. As of August 2017, twelve of the banks in this study had been transferred to a long-term manager, while eight were still being managed by the bank sponsor. I obtained bank permits, monitoring reports, and information about bank status from the US EPA Region 5 Main Office in Chicago, from bank sponsors, and from the online database used by the Corps: RIBITS (Regulatory In-lieu Fee and Bank Information Tracking System) (Corps 2020).

I obtained bank maps from mitigation monitoring reports and used ESRI's ArcMap® Version 10.3.1 software to digitize these maps and to create fifty, randomly generated, potential sample points within the appropriate boundary for each bank. Most of the areas within wetland mitigation banks are jurisdictional wetlands, but banks do contain some areas of upland buffer. Some bank monitoring reports provided a map of delineated wetland boundaries, others provided maps of the mitigation plan (i.e. planned areas of wetland restoration, wetland enhancement, and upland buffer), and some contained only a map of the overall bank boundaries. To ensure that sample points were located within a wetland area, I generated points within delineated wetland boundaries if these maps were available. If I could not obtain a wetland delineation map, then I generated points within the wetland area proposed in mitigation plan maps (excluding areas of upland buffer). I generated sample points within the overall bank boundaries if no other maps were available.

Most of the wetlands restored in mitigation banks in the Chicago District are depressional wetlands dominated by herbaceous species. Banks typically include several specific wetland plant community types, each of which I assigned into one of two broad community types: emergent and wet meadow. My choice of these two community types follows the approach of the Chicago District IRT, which made a distinction in the performance standard thresholds used for these banks between two types of wetland habitat expected in banks: emergent/marsh and sedge meadow/wet prairie (IRT 1997, IRT 2008). At each bank, I attempted to sample one point from each of these community types. I determined the plant community for each point using aerial photos, and confirmed this classification using vegetation and hydrologic conditions observed at the point. I assigned marsh communities with emergent vegetation and evidence of inundation as emergent sites. I classified sedge meadow and wet prairie habitats, which were drier but still appeared to qualify as jurisdictional wetlands, as wet meadow sites. At each bank, I visited points in a randomly assigned order until I could identify one point from each plant community type that met my sampling criteria. I discarded sample points if they were located within 10 meters of the edge of the bank boundary (determined using ArcMap), if they were located in open water, or if they occurred in an area that had clearly not been restored as part of the bank project (e.g. forested areas within the bank boundaries). One bank did not contain any emergent communities, so I sampled a total of 19 emergent points and 20 wet meadow points. I sampled each point once between May and August 2017.

At each sample point I established a 50-meter baseline at a randomly chosen azimuth. I sometimes restricted the baseline azimuth to avoid crossing the bank boundary, developed trails, open water, or crossing between plant community types. After establishing the baseline, I collected data using the wetland ground cover sampling protocol established for the Critical

Trends Assessment Program (CTAP) of the Illinois Natural History Survey (INHS) (Molano-Flores 2002), which has been used to sample existing wetland sites throughout Illinois. Beginning at a randomly selected point on the baseline, I extended a 41-meter transect to the right of, and perpendicular to, the baseline. I placed 20 0.25-m^2 square quadrats, one every two meters, on alternating sides of the transect. If the transect entered open water or a different plant community type, then I truncated the transect and started a new transect from a randomly selected point on the baseline. I repeated this until I established 20 quadrats. I identified and estimated the percent cover of each vascular plant species rooted or floating within each quadrat using the following cover classes: <1%, 1-5%, 5-25%, 25-50%, 50-75%, 75-95%, and 95-100% (Daubenmire 1959). I did not count woody plants greater than one meter tall.

Natural wetland data

I obtained vegetation ground cover data from natural wetlands included in the CTAP database. The CTAP data used in this study were collected throughout Illinois by INHS botanists from 1997 to 2016. CTAP botanists sampled two categories of herbaceous wetlands: randomly selected wetlands and high-quality reference wetlands. Randomly selected sites were chosen from throughout the state based on National Wetlands Inventory data (Molano-Flores 2002). These represent natural wetlands that have experienced levels of degradation that are typical for Illinois. CTAP botanists also collected data from high-quality reference wetlands that were chosen specifically because they have experienced minimal degradation.

I further split the randomly selected CTAP wetlands into groups defined by ecological quality and degree of disturbance. To do this, I used protocol developed for the Illinois Natural Areas Inventory (INAI), a project designed in the 1970's to inventory high-quality natural areas throughout Illinois. A natural quality grading system was developed for the INAI to rank natural areas on a scale of A to E by their successional state and degree of anthropogenic disturbance (White 1978). Sites given a grade of A are stable and have experienced minimal disturbance, while sites given a grade of E are early successional and have been affected by anthropogenic disturbance so severely that the site's original natural community is nearly absent. Each CTAP site was assigned an INAI grade by CTAP botanists, usually during data collection in the field. I used these grades to split randomly selected CTAP wetlands into two groups: random low and random high. I assigned CTAP sites that were given an INAI grade of D or E into the random low group, while I classified sites that were given a grade of A through C as random high. After making these assignments, the random low group did contain a mix of grade D and E sites and the random high group included a mix of grade B and C. I kept the high-quality reference wetlands as their own group, but did observe that these sites consisted entirely of grade A and B wetlands. Therefore, these three groups of CTAP wetlands (random low, random high, and reference) provided a gradient of INAI grades that correlated with natural quality and degree of anthropogenic disturbance.

CTAP botanists recorded a description of the wetland plant community type during sampling visits for each site. I used these descriptions to classify CTAP sites as either emergent or wet meadow communities to match the plant community types I assigned to mitigation bank sites. If the habitat description recorded by CTAP botanists did not fit clearly into one of these two habitat types (e.g. forested floodplains, shrub swamps), then I excluded the site from analysis. Some CTAP sites have been sampled on multiple occasions; in this case I used data from only the most recent sampling visit. To ensure geographic similarity to Chicago District banks, I only used CTAP data from sites in the northern half of Illinois (Figure 3.1).

I compared banks to each natural wetland group, separately for both plant community types, so that banks (number of emergent sites = 19, wet meadow = 20) were compared to 3 categories of natural wetland sites in the CTAP database: random low (emergent = 27, wet meadow = 29), random high (emergent = 26, wet meadow = 16), and reference (emergent = 8, wet meadow = 8).

Data preparation

I identified plant species according to Swink and Wilhelm's (1994) flora specific to the Chicago region. My analysis uses coefficients of conservatism (C-values), a concept that was developed by Swink and Wilhelm to measure and evaluate the floristic quality of plant species and communities. C-values were subjectively assigned by the authors, botanists with substantial knowledge of northeastern Illinois' flora, to each native plant species in the region. These values were designed to rate a species' tolerance to anthropogenic disturbance and fidelity to natural communities on a scale from 0 to 10 (Swink and Wilhelm 1994). Species given a score of 10 are conservative species only found in undegraded natural communities, while species with a score of 0 are highly tolerant of anthropogenic disturbance and may be found growing both in degraded and high-quality communities. I obtained the C-value and native status for every species in this study from Swink and Wilhelm's flora (1994), except for the common species *Phragmites australis, Typha angustifolia*, and *Typha* x *glauca*. These are listed as native species with a C-value of 1 in Swink and Wilhelm's 1994 flora; however, I treated them as non-native species to reflect changes made in a more recent, authoritative, regional flora (Wilhelm and Rericha 2017).

Numerical analysis

I compared sites using several univariate, vegetation-based metrics that reflect performance standards that were developed by the Chicago District IRT to evaluate plant communities in the banks included in this study (IRT 1997, IRT 2008). These included two floristic quality indices calculated using species C-values: native mean C and native Floristic Quality Index (native FQI). Native mean C is the average of the C-values for all the native plant species found at a site. Native FQI incorporates native species richness with floristic quality, and is calculated as:

$$\text{native FQI} = \text{native mean C} \times \sqrt{S}$$

S = the total number of native plant species at a site

For each bank and CTAP site I calculated native species richness, native mean C, native FQI, and relative importance value of native species. Following the method used by the IRT and bank sponsors (IRT 1997, IRT 2008), I calculated native relative importance value by determining the cumulative relative cover and relative frequency for all native species at a site, and taking the average of these values.

I compared univariate vegetation metrics among site types (banks, random low, random high, and reference) separately for emergent and wet meadow habitats. I used a single factor analysis of variance (ANOVA) to produce parameter estimates for the mean metric score for each group and to test for an overall difference among group means. I examined plots of residuals to determine if data met assumptions of equal variance between groups and normally distributed residuals; data met these assumptions for all metrics. If an ANOVA showed a significant overall difference among groups ($\alpha = 0.05$), then I conducted three pairwise comparisons (between banks and each of the three natural wetland categories) using Tukey's

Honest Significant Difference (HSD) test to control for the experimentwise error rate. For all tests, the null hypothesis was that there was no difference between group means. I conducted all statistical tests using R version 3.4.3 (R Core Team 2017).

To assess the relationship between bank age (the time elapsed between site construction and my 2017 sampling) and the selected vegetation metrics, I conducted linear regression analysis for each habitat type, using bank age as the only explanatory variable. I conducted these tests separately for each plant community type.

I used non-metric multidimensional scaling (NMDS) to visualize differences in community composition between sites for both emergent and wet meadow habitats. I conducted NMDS using the Morisita-Horn index to measure pairwise dissimilarity between sites, based on the square-root transformed relative cover data for each species at each site. The Morisita-Horn index is appropriate for species abundance data and is especially sensitive to the most abundant species (Jost et al. 2011). I performed the NMDS using a maximum of 50 random starts to reach a stable solution. Using a scree plot to evaluate the stress values generated for the NMDS conducted with different numbers of axes, I determined to use three dimensions both for emergent (stress = 0.15) and wet meadow (stress = 0.14) habitats. I plotted all sites in NMDS space for each community type. To improve my interpretation of these graphs, I also plotted the NMDS location of the five plant species with the highest relative cover in each habitat type. I conducted this procedure using the *metaMDS* function in the vegan package in R (Oksanen et al. 2017).

I tested for statistical differences in community composition between site types using permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001). I tested for overall and pairwise differences between groups using the *adonis* function in the vegan package in R (Oksanen et al. 2017). I performed tests on the square-root transformed relative cover data, using the Morista-Horn index, with 9999 permutations. The reliability of PERMANOVA tests is compromised when the dispersion of multivariate dissimilarity points is not equal among groups (Anderson and Walsh 2013), so I tested to ensure that groups had equal dispersion. If the PERMANOVA test results indicated a significant overall difference between groups, I used the *adonis* function to make pairwise comparisons between banks and each of the three natural wetland groups. I used Bonferroni-corrected alpha values ($\alpha = 0.0125$) to evaluate each pairwise comparison to maintain an experimentwise error rate of $\alpha = 0.05$.

3.3 Results

Univariate vegetation-based metrics

Across all metrics, and both habitat types, mean vegetation metric scores always increased from random low, to random high, to reference sites, confirming that these three groups of natural wetlands provide a meaningful gradient of ecological integrity against which to measure banks (Table 3.1). The ANOVA results revealed a significant overall difference ($P < 0.05$) between site types for each of the four univariate metrics I tested, for both emergent and wet meadow habitats (Table 3.2).

Banks scored higher than random low CTAP sites for every vegetation metric (Figures 3.2-3.5, Table 3.1), though this difference was not statistically significant for native species richness and native relative importance value in emergent sites (Table 3.2). For native relative importance value, particularly in emergent sites (Figure 3.5a), scores for banks and random low sites were highly variable, and both of these groups contained sites that were dominated by either native or non-native species.

Mean vegetation metric scores were lower in banks than in random high sites across all metrics (Figures 3.2-3.5, Table 3.1), but the size of these differences varied. In emergent sites the differences in mean scores between these groups were not statistically significant for any of the metrics I tested (Table 3.2). In wet meadow sites the random high group scored significantly higher than banks for only native FQI (P = 0.0376) (Figure 3.4b, Table 3.2), though the results for native mean C are worth noting as well (P = 0.0869, Figure 3.3b, Table 3.2).

Reference sites scored higher than banks for all metrics in both plant community types (Figures 3.2-3.5, Table 3.1), and I found this difference to be statistically significant in every case (Table 3.2) except for native relative importance value in wet meadow sites, which only slightly exceeded my chosen significance threshold (P = 0.0584, Figure 3.5b, Table 3.2). Reference site data often had lower variance than other site types, particularly for native mean C and native relative importance value, likely because my sample size for reference sites was smaller than the other groups.

Bank Age

I found evidence of a significant negative relationship between bank age and native relative importance value (Table 3.3). This relationship was statistically significant for emergent sites (Slope = -0.041, SE = 0.017, F = 6.052, P = 0.025, R^2 = 0.263), and was noteworthy, but not below my chosen significance threshold, for wet meadow sites (Slope = -0.022, SE = 0.012, F = 3.689, P = 0.071, R^2 = 0.170). There was no significant relationship between bank age and any of the other vegetation-based metrics I tested.

Community composition

NMDS plots differentiated between the plant community composition of banks and natural wetland sites. For emergent habitats, banks are concentrated in the lower middle portion of the plot of NMDS axes 1 and 2 (Figure 3.6). NMDS axis 1 seems to represent a gradient of floristic quality, disturbance, and possibly hydrology, with random low and some random high sites associating with the non-native *Phalaris arundinacea* on the left, reference and some random high sites associating with the native aquatic species *Lemna minor* on the right, and banks occupying a position between random low and reference sites. Banks show stronger separation from all natural wetland groups along axis 2, associating with the non-native species *Typha angustifolia* and *Phragmites australis*. A clear relationship between banks and natural wetland sites is not evident along NMDS axis 3 (Figure 3.7).

NMDS plots for wet meadow sites show similar evidence for relationships in community composition among different site types. More clearly than for emergent sites, NMDS axis 1 seems to correlate with disturbance and floristic quality, with random low sites associating with *Phalaris arundinacea* at lower values, reference and random high sites associating with the native *Carex stricta*, *Calamagrostis canadensis*, and *Solidago gigantea* at higher values, and most banks occupying an intermediate position (Figure 3.8). Once again, banks show a strong separation from all natural wetland sites along axis 2. However, there are several bank sites in Figure 3.8 that associate with random low sites and *Phalaris arundinacea*, and several other banks that seem more similar to reference sites. Trends in bank position relative to other groups are less clear along NMDS axis 3 (Figure 3.9).

The results of PERMANOVA tests for differences in community composition showed a significant overall difference between site types and significant pairwise differences between banks and random low, random high, and reference sites for both community types (Table 3.4).

Dominant species

The species with the highest average relative cover are listed for each site type in Table 3.5. The three species with the highest average cover in emergent bank sites (*Typha angustifolia, Phragmites australis, Phalaris arundinacea*) are all non-native, and have a cumulative average cover of 49.08%. In wet meadow bank sites, the three most dominant species (*Phalaris arundinacea, Solidago canadensis,* and *Silphium perfoliatum*) have a cumulative average cover of 37.08%. *Phalaris arundinacea* shows a high relative cover in natural wetlands as well as banks, especially in random low sites, where its relative cover is much higher than that found in banks for both community types. *Typha angustifolia* and *Phragmites australis* are among the most dominant species in randomly selected natural emergent wetlands, but their relative cover is higher in banks than in any other site type. Only the reference groups do not include a non-native species among their five most dominant species.

3.4 Discussion

Vegetation metrics

My primary objective in this study was to determine the ecological condition of wetland mitigation banks relative to natural wetlands, based on several measurable characteristics of their plant communities. My results indicate, consistently across several metrics, that the condition of plant communities in banks exceeds that of low-quality natural wetlands, but falls short of high-quality, undegraded reference wetlands.

I found that native species richness in banks was higher than that in random low sites, and lower than in reference wetlands. Previous studies have documented mixed results when comparing richness between restored and natural wetlands. Some have found that native and total species richness in restored wetlands may match or exceed the values observed both in typical natural wetlands (Gutrich et al. 2009, Matthews et al. 2009) and even in high-quality reference wetlands (Hopple and Craft 2013), but in other cases restorations fall short of, and are unlikely ever to reach, the richness in reference wetlands (Campbell et al. 2002, Gutrich et al. 2009). Average species richness across wetland mitigation banks may be less than that of reference wetlands, but some bank sites have shown the ability to reach richness levels similar to that of reference sites (Stefanik and Mitsch 2012). Planting and seeding of native species during site construction may be necessary for restored wetlands to possess species richness equivalent to that of natural wetlands (Wall and Stevens 2015) and these activities likely influenced the results I observed in mitigation banks. All the banks included in my study were planted or seeded with native species during initial construction, which was a requirement of their permit agreements. Many also continued to receive additional planted and seeded species throughout their 5-year management periods, which likely increased their native species richness values relative to natural wetlands.

Temporal fluctuations in species richness complicate efforts to evaluate this metric in restored wetlands. Species richness in wetland mitigation banks has been shown to change significantly, and sometimes erratically, over time in wetland mitigation banks that are less than 5 years old, likely due to the disturbance caused by site construction and the input of planted and seeded species (Spieles 2005, Spieles et al. 2006). Species richness is affected by successional patterns, as the plant communities in restored wetlands, influenced early on by rapidly-colonizing, annual, and ruderal species, eventually experience species turnover and increasing dominance by long-lived perennial species (Matthews and Endress 2010, Stefanik and Mitsch 2012). However, I found no evidence for a relationship between bank age and native species richness in this study. Such a relationship may have indicated temporal shifts in the species richness in banks. Sampling only banks that were more than eight years old increased the likelihood that initial fluctuations in species richness had diminished, and that the native richness values I observed reflected plant communities that had reached some level of stability, rather than communities in which high diversity was temporarily influenced by the input of new species through management effort or by rapid succession. However, others have reported that species richness in restored wetlands may continue to shift, in either direction, for more than a decade following restoration (Mulhouse and Galatowitsch 2003, Aronson and Galatowitsch 2008, Gutrich et al. 2009), even showing significant year-to-year changes in the number and composition of species observed (Wall and Stevens 2015). Even in older wetland mitigation banks, native species richness may still be experiencing these temporal shifts.

The native mean C values I reported from banks also support the conclusion that banks have higher ecological integrity than low-quality natural wetlands but fail to reach that of high-quality reference sites. Native mean C scores in mitigation bank wetlands may be low relative to reference wetlands if they are unable to develop plant communities that include the full suite of species found in natural wetlands. Aronson and Galatowitsch (2008) found that uncommon species may be poorly represented in restored wetlands, as 70% of infrequent species, and 93% of the rare species found in equivalent natural wetlands were never observed in restored wetlands. The difficulty of establishing rare species, which are generally more likely to have high C-values, in restored wetlands may explain the difference I observed between native mean C values in banks and reference wetlands. Others have previously found that native mean C values in restored wetlands are lower than those in high-quality reference wetlands (Hopple and Craft 2013), and in natural wetlands of typical condition (Matthews et al. 2009, Van den Bosch and Matthews 2017). My study supports these results in part, but also provides evidence that banks have higher native mean C scores than low-quality natural wetlands. One factor that may increase the native mean C values of banks relative to low-quality natural sites is the heavy planting and seeding effort made at banks, as the introduction of species with high C-values in created wetlands can elevate floristic quality measures (DeBerry and Perry 2015).

Native FQI is derived from native species richness and native mean C, and so it reflects the same trend as both of these metrics, with values in banks that are higher than low-quality natural wetlands but lower than those in reference wetlands. Others have reported that FQI in restored wetlands may very quickly become even with, or even higher than, the FQI measured in natural wetlands (Matthews et al. 2009, Hopple and Craft 2013, Van den Bosch and Matthews 2017). However, Stefanik and Mitsch (2012) found that average FQI in wetland mitigation banks was less than that in reference wetlands, and that older banks, in particular, had lower FQI values. Like native species richness and native mean C, I suggest that the lower native FQI values I reported in banks compared to high-quality reference wetlands indicate that the plant communities in banks have not been able to accumulate species assemblages that include many of the uncommon, high C-value species present in reference wetlands.

Non-native species dominance

Dominance by non-native species is a critical challenge to the ecological success of restored wetlands. Herbaceous wetlands in the interior plains of the United States, including the Chicago District of the Corps, experience a large amount of pressure from non-native plants (EPA 2016). Studies of wetland restoration in the region have found that restored wetlands are dominated by non-native species to a greater degree than are natural wetlands (Aronson and Galatowitsch 2008, Matthews and Spyreas 2010) and that mitigation wetlands often do not meet regulatory performance standards that require project managers to keep non-native dominance below a minimum threshold (Matthews and Endress 2008, Van den Bosch and Matthews 2017). The influence of non-native species, as interpreted through the native relative importance value metric which represents the inverse of non-native species dominance, is evident in the banks I assessed as well. Bank condition relative to natural wetland groups was similar for this metric as it was for the other metrics I tested, with banks generally scoring higher than low-quality natural wetlands and falling short of high-quality reference sites, but it is also helpful to consider the absolute value of this measure in banks itself. The estimated mean native relative importance values in emergent (mean = 0.52, SE = 0.07) and wet meadow (mean = 0.71, SE = 0.05) bank sites indicate that the relative cover and abundance of non-native species were high: accounting for nearly 50% of the importance value in emergent bank wetlands and nearly 30% in wet meadow banks wetlands. These values for non-native species dominance exceed what I found in high-quality natural wetlands and greatly exceed the thresholds for this metric that were established by the IRT for the ecological performance standards applied to these bank sites. To meet this performance standard, the banks in my study were required to show a cumulative relative importance value of less than 0.05 for all non-native and weedy species at the end of their five year monitoring period (IRT 1997, IRT 2008), though this threshold has been increased to 0.10 in a more recent version of the Chicago District performance standards (IRT 2017).

Much of the influence of non-native species on wetland mitigation banks in my study can be attributed to three species: *Phalaris arundinacea*, *Typha angustifolia*, and *Phragmites australis* (Table 3.5). In wetland restorations in the Midwest, these species, particularly *Phalaris arundinacea*, can increase in abundance more quickly than native species, become dominant in the plant communities of restored wetlands, and contribute to increasing dissimilarity between restored wetlands and high-quality reference sites over time (Mulhouse and Galatowitsch 2003, Aronson and Galatowitsch 2008, Matthews and Spyreas 2010). Increasing *Phalaris arundinacea* dominance in restored wetlands may be associated with low measures of species richness and floristic quality (Spyreas et al. 2010) and may increase biotic homogenization, prevent the establishment of native species that are inefficient colonizers, and cause the loss of native species with small population sizes (Aronson and Galatowitsch 2008). Non-compliance with performance standards in mitigation wetlands can also be attributed largely to *Phalaris arundinacea* and *Typha angustifolia* (Matthews and Endress 2008, Van den Bosch and Matthews 2017).

A regional trend of increasing dominance by these common non-native species may explain the negative relationship I have reported between native relative importance value and age since restoration in mitigation bank wetlands; however, a time series analysis including vegetation data from multiple sampling visits across several years at bank sites would be necessary to evaluate this hypothesis. Another possible explanation for this trend is that wetland restoration techniques have improved over the last two decades, so that younger banks are more resistant to invasion by non-native species. In a study of wetland mitigation banks in Ohio, Stefanik and Mitsch (2012) found evidence that younger banks may have been built with better restoration methods, and possessed greater species richness and floristic quality than older sites. It is possible that such an improvement in wetland mitigation practice and management has occurred in the Chicago District as well. Bank monitoring reports indicate that bank sponsors conducted extensive management of *Phalaris arundinacea*, *Typha angustifolia*, and *Phragmites australis* during the management periods at banks, though it is not clear if more resources were dedicated to this task at banks that were constructed more recently. Both *Typha angustifolia* and *Phragmites australis* were listed as native species in the regional flora (Swink and Wilhelm

1994) that was used for species identification during the management and monitoring period for most of the banks in my study, though they are now considered as non-native (Wilhelm and Rericha 2017). While the performance standards assigned to these banks have always restricted these as undesirable species, it is plausible that changing understanding about the provenance of these species has led to an increase in the efforts made to control them in restored wetlands.

Community composition

Another objective of my study was to compare differences in plant community composition among wetland mitigation banks and different groups of natural wetlands. The significant differences I reported between the plant communities in banks and those in random low, random high, and reference wetlands can be explained in part by patterns of dominance by *Phalaris arundinacea, Typha angustifolia,* and *Phragmites australis*. The Morisita-Horn index I used to calculate dissimilarity in community composition between sites is especially sensitive to the most abundant species, and so was well-suited for assessing the effect of these abundant non-native species.

The relationship between community composition in banks and natural wetland sites in my study showed a striking degree of consistency between emergent and wet meadow sites. For both community types, banks occupied a clearly intermediate position between random low and reference natural wetlands along an axis associated with site degradedness, with random low wetlands and a few banks characterized by *Phalaris arundinacea,* and reference sites and a few high-performing banks associated with an abundant native species. The relative cover of *Phalaris arundinacea* alone generally follows this relationship between banks and natural wetland sites as well, with random low sites showing especially high relative cover values of more than 40% for *Phalaris arundinacea* in both community types (Table 3.5). In a study including many of the same CTAP wetlands I have assessed here, Price et al. (2017) found that *Phalaris arundinacea* was increasing in natural wetlands in Illinois, causing regional homogenization and loss of beta diversity of native species. At the time of my monitoring visits, banks contained less relative cover of *Phalaris arundinacea* than the lowest quality natural wetlands, but the regional trend of increasing dominance by this species is concerning for banks that already have greater abundance of this species than do high-quality natural wetlands.

My second conclusion from community composition analysis is that banks clearly separate from all natural wetland groups along a second axis in NMDS space. In emergent sites, this difference is clearly characterized by *Typha angustifolia* and *Phragmites australis*, the two species with the highest relative cover in emergent bank sites (Table 3.5). *Typha angustifolia* has been reported as a problematic invader in other Illinois mitigation wetlands (Matthews and Endress 2008, Van den Bosch and Matthews 2017), while some natural wetlands in Illinois have also appeared to be converging towards a state of *Phragmites australis* dominance (Price et al. 2017). These species are characteristic of marshes, roadside ditches, and lake borders around Chicago, and can be tolerant of alkaline and disturbed wetlands (Swink and Wilhelm 1994). In the past, there has been evidence that constructed mitigation wetlands possessed hydrologic regimes that were more permanently wet, sometimes including areas of open water, than is characteristic of natural wetlands (Cole and Brooks 2000). While the creation of hydrologically appropriate mitigation wetlands may have improved since that time, it is possible that the older mitigation banks in my study included a large amount of open water wetlands, which would provide favorable habitat for lake edge species such as *Typha angustifolia* and *Phragmites australis*. Wet meadow sites in banks also showed a separation from all natural wetlands along a second NMDS axis, but the species driving this distance are less clear.

Conclusion

I have examined wetland mitigation banks that have aged past their required five-year management and monitoring periods. Once this period is complete, bank sponsors must transfer bank properties to a long-term owner and manager, and provide financial, legal, and planning resources for long-term site management (Corps and EPA 2008). At the time I collected data from banks, twelve had been transferred to long-term owners such as the Illinois Department of Natural Resources, county forest preserve districts, and local townships; however, eight banks were still held and managed by the original bank sponsor. Bank sponsors typically conducted extensive management activities during the initial monitoring period, including non-native species control, seeding or planting of native species, hydrologic alteration, and prescribed burning. It is likely that some of these activities have continued at banks following their transfer to a long-term site owner; however, I do not have data documenting ongoing management at the banks in my study. The ecological condition of banks I have observed has likely been significantly influenced by the type and amount of ecological management that has occurred since the completion of their five-year monitoring periods. Future work in this area could assess the long-term management that is conducted at mitigation banks to determine if it has occurred and what has been its effect on the ecological condition of banks.

Entrepreneurial wetland mitigation banking first developed in the United States in the early 1990’s (Hough and Robertson 2009) but has seen large increases in its use much more recently (IWR 2015, Hough and Harrington 2019). This has limited the number and age of banks available for independent evaluation, so that studies assessing the condition of plant communities in banks have largely been able to sample only a small number of relatively young banks. In this study, I have conducted one of the most thorough examinations of the ecological results of wetland mitigation banks to date. The well-developed banking market in the Chicago District allowed me to sample a relatively large number of sites that included some of the oldest private mitigation banks in the United States, so that my results can expand the generality and temporal scope of previous studies. In addition, by comparing banks to groups of natural wetlands of varying floristic quality, I have been able to provide precise information about bank condition relative to natural wetlands. This information can give regulatory agencies, bank sponsors, and wetland restoration practitioners a clearer picture of what has been the ecological outcome of mitigation banking in its first three decades. This will help those involved with wetland mitigation to assess if wetland banks are facilitating the achievement of no net loss of wetland resources, and can inform future goal setting, bank management, and policy development.

3.5 Figures

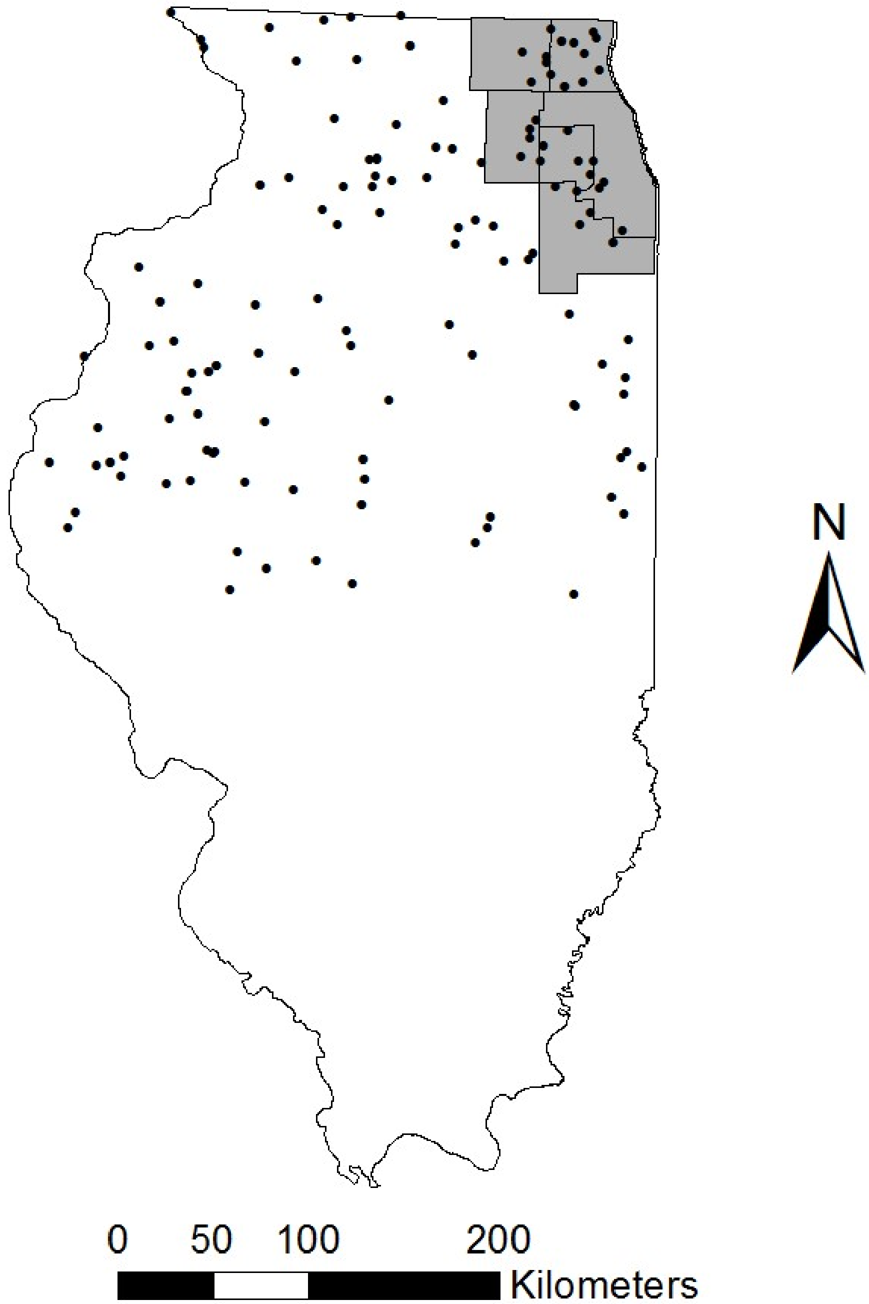

Figure 3.1. Area regulated by the Chicago District of the Army Corps of Engineers, within which wetland mitigation bank sites were located (shaded counties), and locations of CTAP sites used in this study (dark circles).

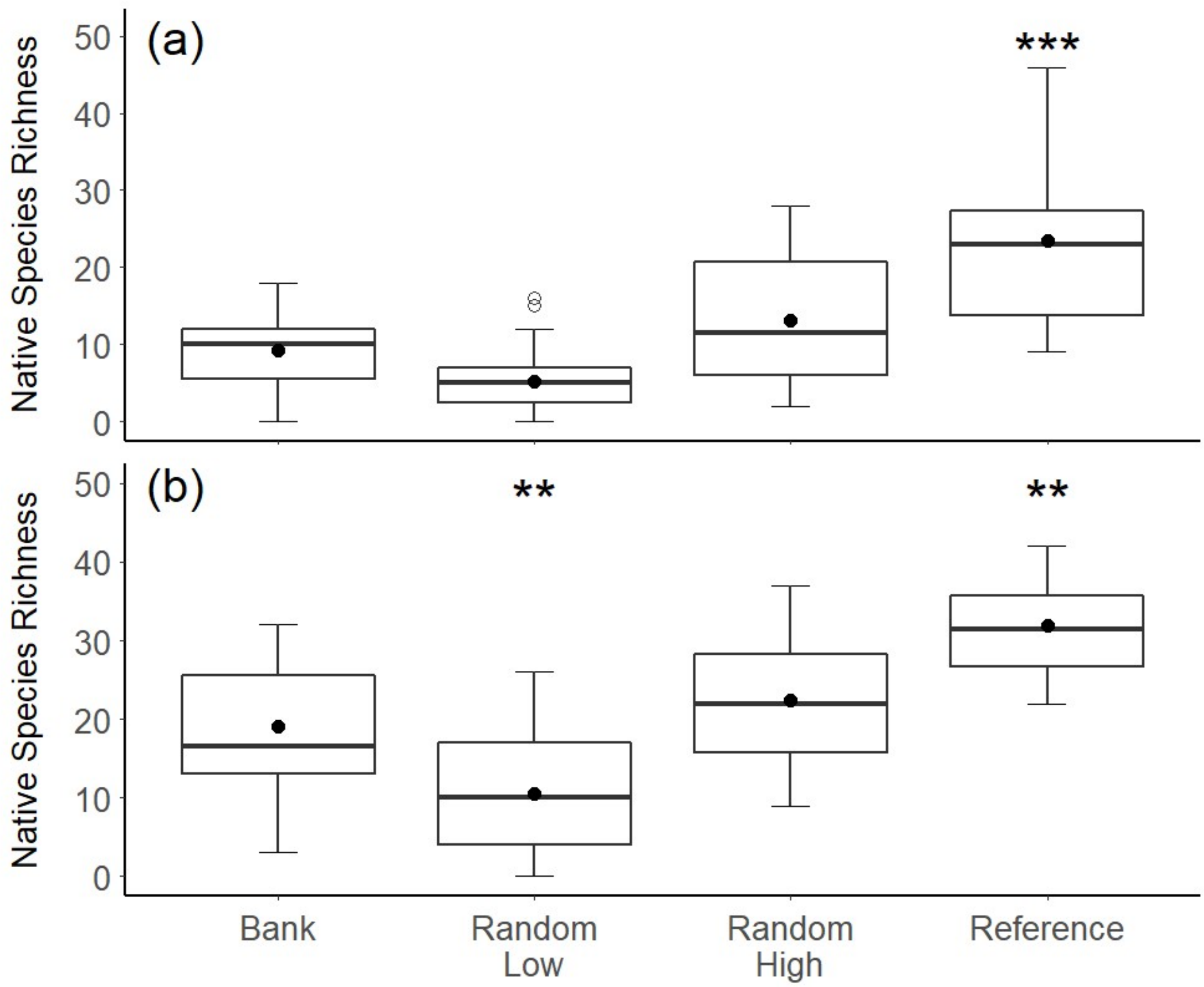

Figure 3.2. Box-and-whisker plots illustrating the distribution of native species richness values by wetland site type for (a) emergent and (b) wet meadow habitats. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). Asterisks above box-and-whisker plots indicate if the individual natural wetland groups were significantly different from banks (*** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$) based on Tukey's Honest Significant Difference (HSD) test.

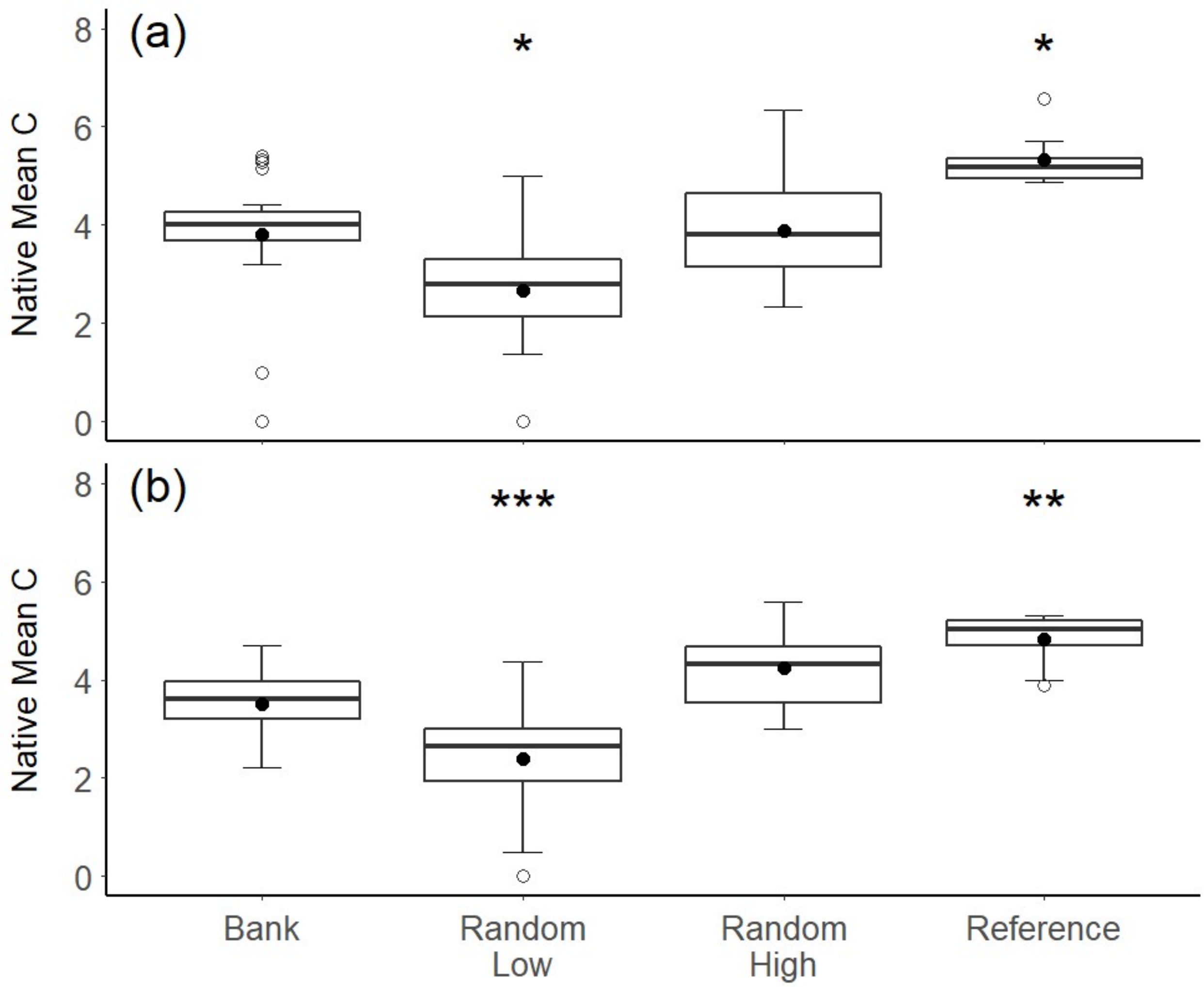

Figure 3.3. Box-and-whisker plots illustrating the distribution of native mean C values by wetland site type for (a) emergent and (b) wet meadow habitats. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). Asterisks above box-and-whisker plots indicate if the individual natural wetland groups were significantly different from banks (*** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$) based on Tukey's Honest Significant Difference (HSD) test.

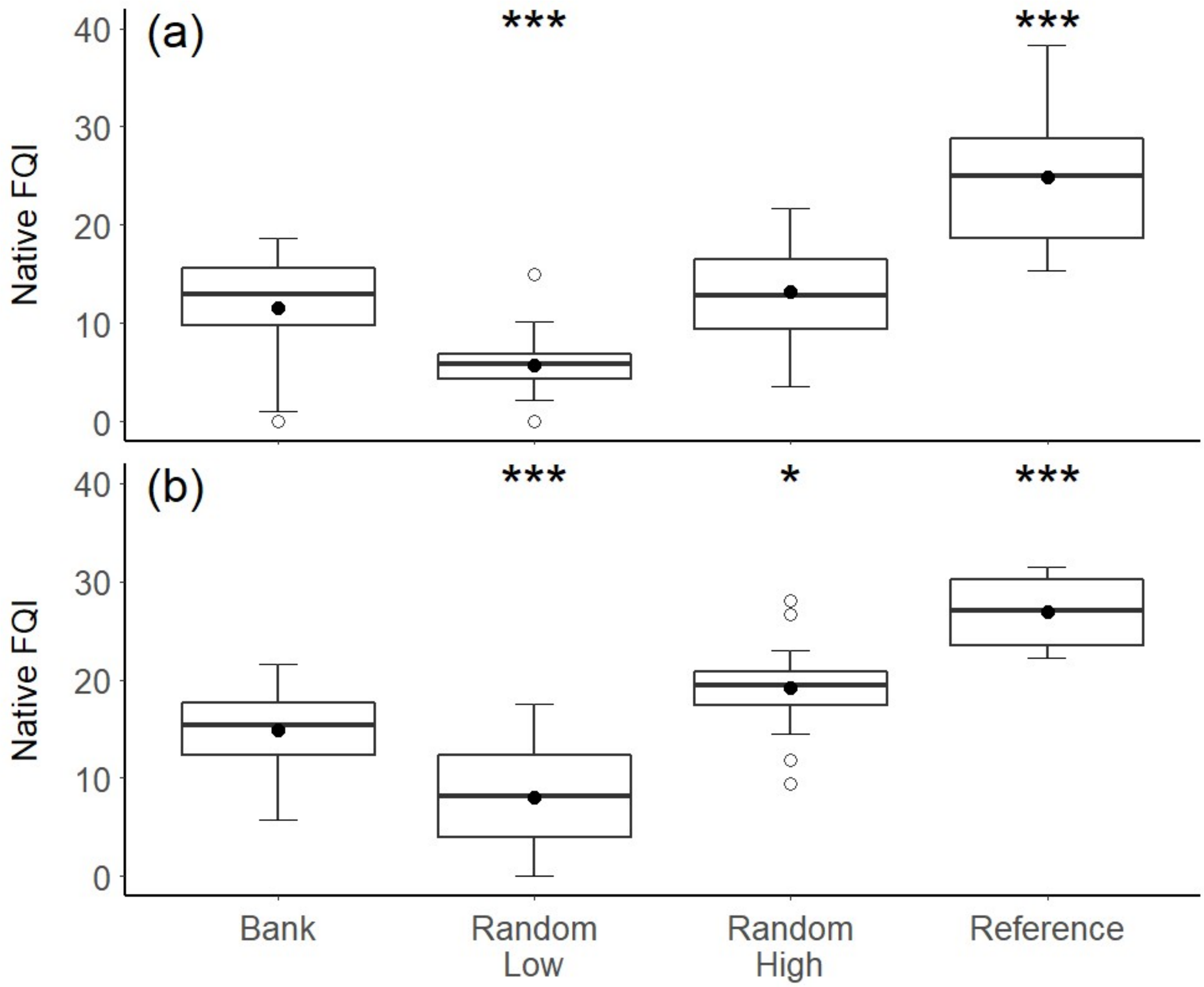

Figure 3.4. Box-and-whisker plots illustrating the distribution of native FQI values by wetland site type for (a) emergent and (b) wet meadow habitats. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). Asterisks above box-and-whisker plots indicate if the individual natural wetland groups were significantly different from banks (*** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$) based on Tukey's Honest Significant Difference (HSD) test.

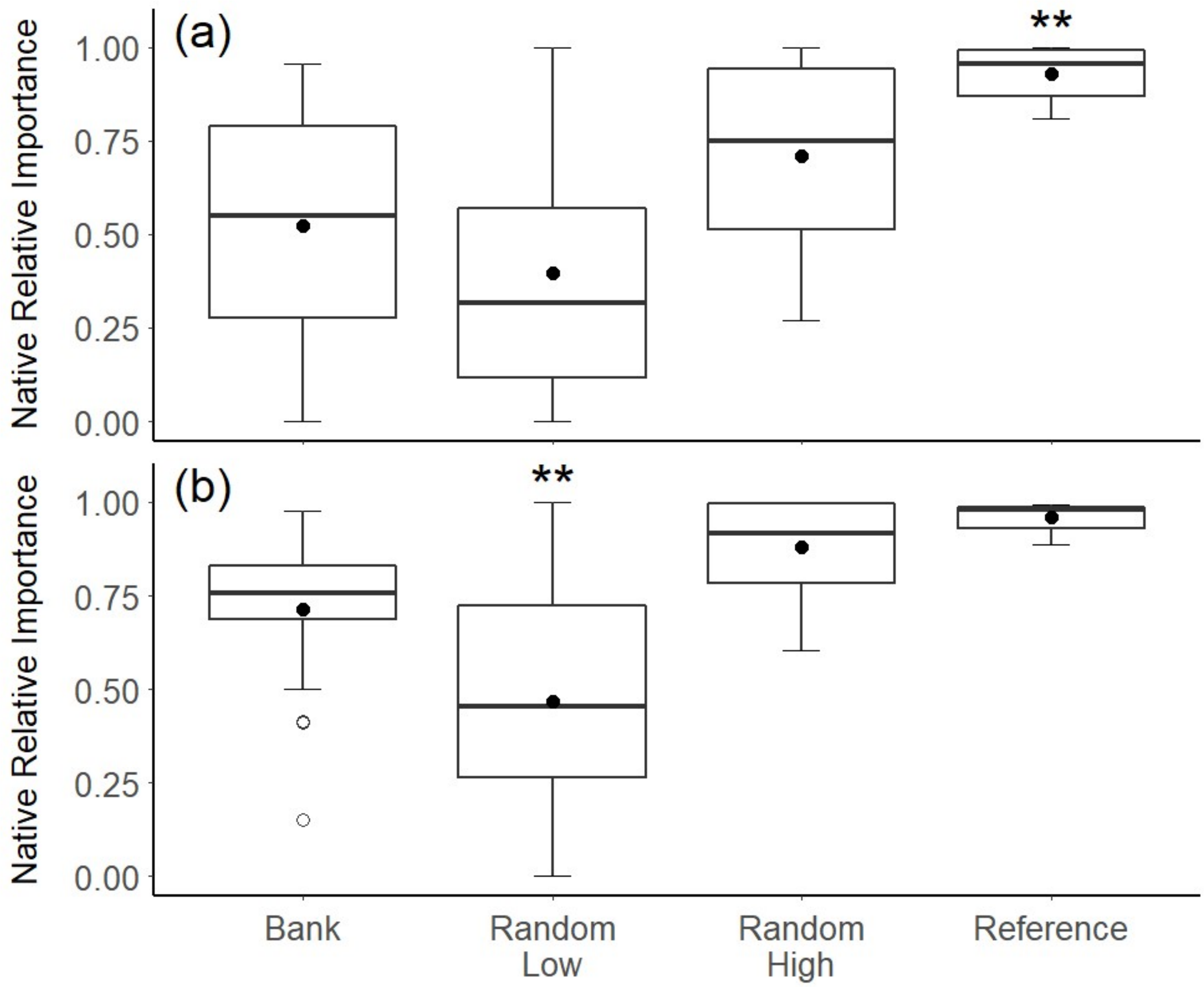

Figure 3.5. Box-and-whisker plots illustrating the distribution of native relative importance values by wetland site type for (a) emergent and (b) wet meadow habitats. Box-and-whisker plots show the mean (dark circles), median (line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). Asterisks above box-and-whisker plots indicate if the individual natural wetland groups were significantly different from banks (*** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$) based on Tukey's Honest Significant Difference (HSD) test.

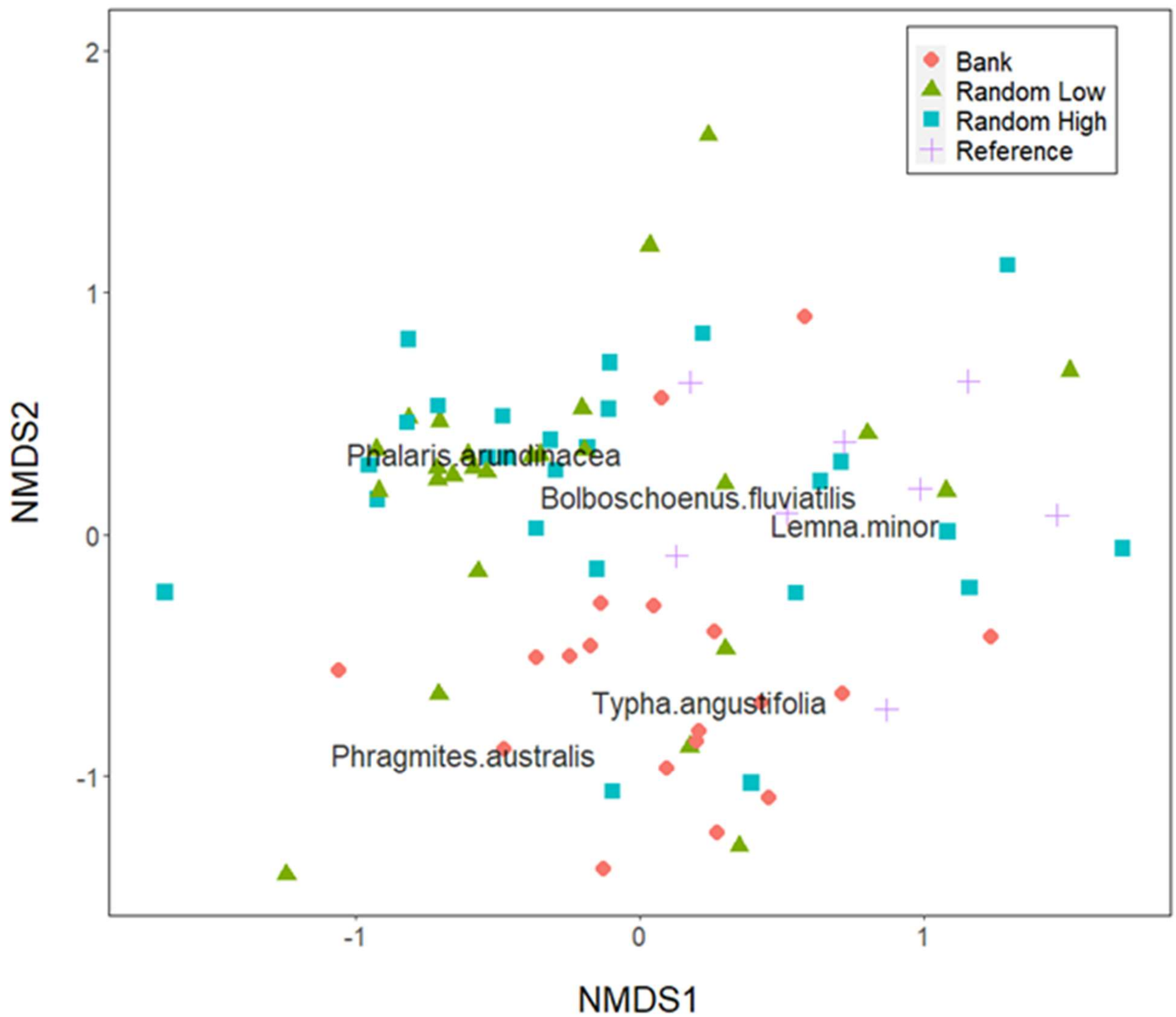

Figure 3.6. Non-metric multidimensional scaling (NMDS) plot for emergent sites, based on the Morisita-Horn dissimilarity index, and calculated using plant species relative cover data treated with a square-root transformation. The plot shows different wetland site types, as well as the five species with the highest relative cover across all emergent sites. Plot shows NMDS axes 1 and 2 (stress = 0.15).

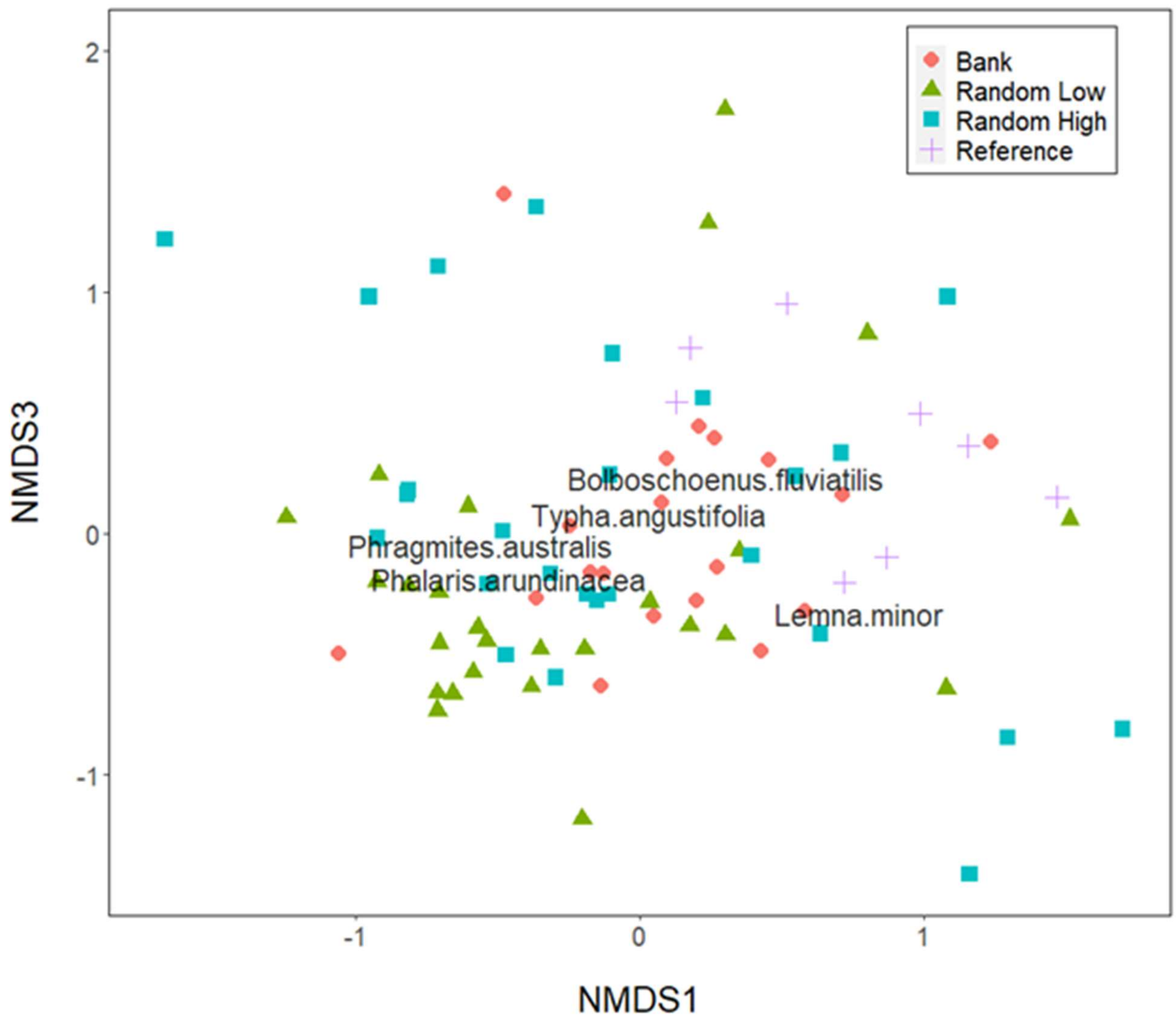

Figure 3.7. Non-metric multidimensional scaling (NMDS) plot for emergent sites, based on the Morisita-Horn dissimilarity index, and calculated using plant species relative cover data treated with a square-root transformation. The plot shows different wetland site types, as well as the five species with the highest relative cover across all emergent sites. Plot shows NMDS axes 1 and 3 (stress = 0.15).

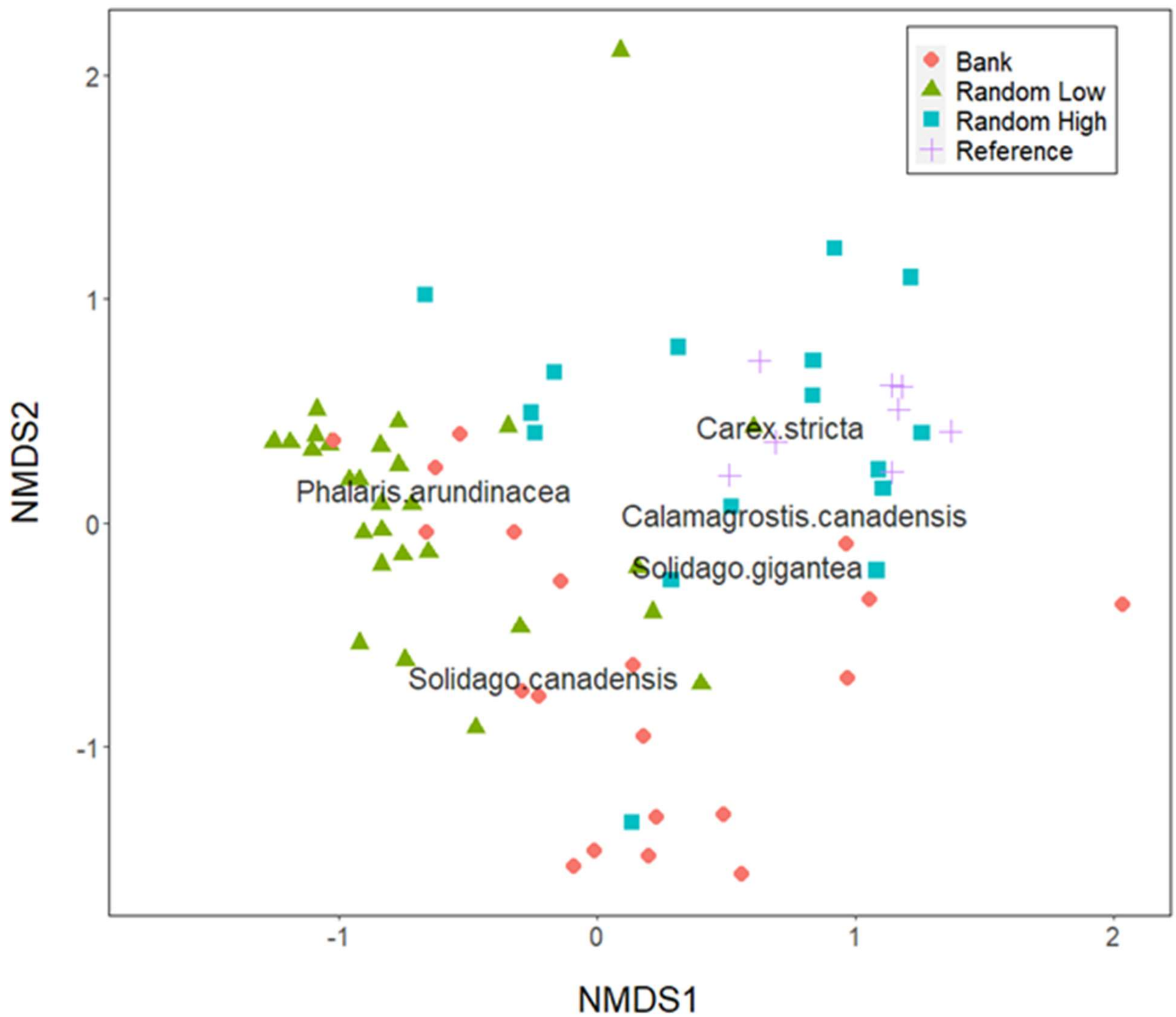

Figure 3.8. Non-metric multidimensional scaling (NMDS) plot for wet meadow sites, based on the Morisita-Horn dissimilarity index, and calculated using plant species relative cover data treated with a square-root transformation. The plot shows different wetland site types, as well as the five species with the highest relative cover across all wet meadow sites. Plot shows NMDS axes 1 and 2 (stress = 0.14).

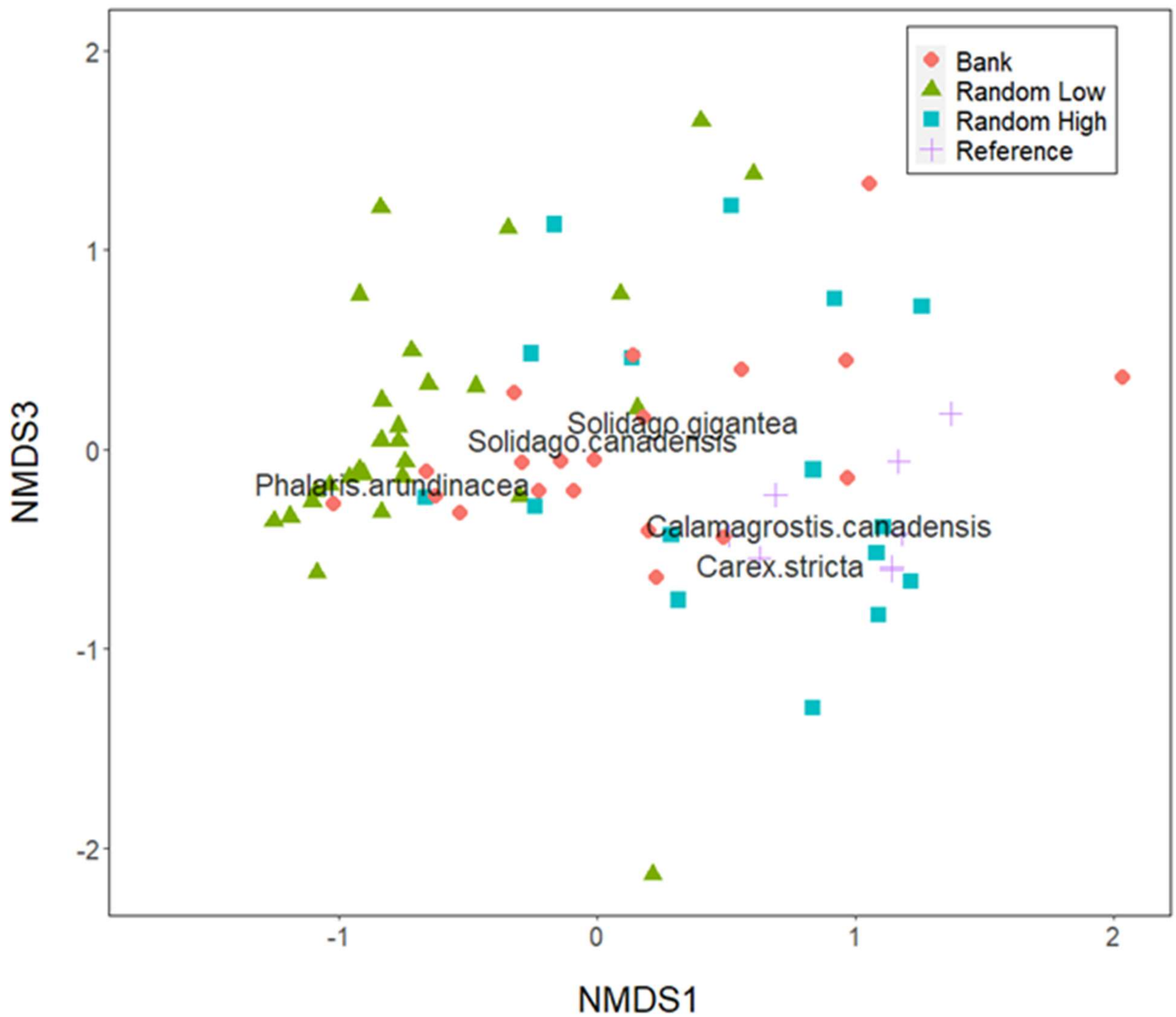

Figure 3.9. Non-metric multidimensional scaling (NMDS) plot for wet meadow sites, based on the Morisita-Horn dissimilarity index, and calculated using plant species relative cover data treated with a square-root transformation. The plot shows different wetland site types, as well as the five species with the highest relative cover across all wet meadow sites. Plot shows NMDS axes 1 and 3 (stress = 0.14).

3.6 Tables

Table 3.1. Parameter estimates for group means and standard errors resulting from ANOVA comparisons between wetland site types for target vegetation metrics. Tests were conducted separately for emergent and wet meadow sites.

		Emergent		Wet Meadow	
Metric	Site Type	Mean	*SE*	Mean	*SE*
Native Species Richness	Banks	9.21	1.58	19.00	1.84
	Random Low	5.22	2.07	10.59	2.39
	Random High	13.08	2.08	22.38	2.75
	Reference	23.38	2.91	31.88	3.44
Native Mean C	Banks	3.81	0.27	3.51	0.21
	Random Low	2.67	0.35	2.39	0.27
	Random High	3.89	0.36	4.25	0.31
	Reference	5.33	0.50	4.83	0.38
Native FQI	Banks	11.58	1.10	14.85	1.04
	Random Low	5.73	1.44	8.02	1.36
	Random High	13.14	1.45	19.15	1.56
	Reference	24.85	2.03	26.90	1.95
Native Relative Importance Value	Banks	0.52	0.07	0.71	0.05
	Random Low	0.39	0.09	0.47	0.07
	Random High	0.71	0.09	0.88	0.08
	Reference	0.93	0.12	0.96	0.10

Table 3.2. Test statistics and significance values for hypothesis tests conducted to determine differences between wetland site types. Overall comparisons were conducted using an ANOVA and specific post hoc comparisons between banks and each natural wetland site type were made using Tukey's Honest Significant Difference (HSD) test to control for the experiment wise error rate. Tests were conducted separately for emergent and wet meadow sites.

		Emergent		Wet Meadow	
Metric	Comparison	*F*	*P*	*F*	*P*
Native Species Richness	Overall	15.97	<0.0001	17.10	<0.0001
	Banks to Random Low	-	0.2243	-	0.0041
	Banks to Random High	-	0.2556	-	0.6129
	Banks to Reference	-	<0.0001	-	0.0020
Native Mean C	Overall	11.90	<0.0001	22.68	<0.0001
	Banks to Random Low	-	0.0105	-	0.0005
	Banks to Random High	-	0.9953	-	0.0869
	Banks to Reference	-	0.0168	-	0.0054
Native FQI	Overall	34.50	<0.0001	42.85	<0.0001
	Banks to Random Low	-	0.0007	-	<0.0001
	Banks to Random High	-	0.7057	-	0.0376
	Banks to Reference	-	<0.0001	-	<0.0001
Native Relative Importance Value	Overall	9.46	<0.0001	16.60	<0.0001
	Banks to Random Low	-	0.4625	-	0.0023
	Banks to Random High	-	0.1542	-	0.1437
	Banks to Reference	-	0.0074	-	0.0584

Table 3.3. Estimated slopes, standard errors, test statistics, significance values, and multiple R^2 values for linear regression models fitting vegetation metrics against bank age. Analyses were conducted separately for emergent (n=19) and wet meadow (n=20) plant communities.

	Emergent					Wet Meadow				
Metric	Slope	*SE*	*F*	*P*	R^2	Slope	*SE*	*F*	*P*	R^2
Native Species Richness	-0.17	0.32	0.27	0.6118	0.02	0.04	0.51	0.01	0.9394	0.00
Native Mean C	-0.04	0.08	0.25	0.6234	0.01	-0.02	0.04	0.38	0.5462	0.02
Native FQI	-0.20	0.31	0.40	0.5378	0.02	-0.10	0.27	0.15	0.7056	0.01
Native Relative Importance	-0.04	0.02	6.05	0.0249	0.26	-0.02	0.01	3.69	0.0707	0.17

Table 3.4. Test statistics from PERMANOVA tests comparing site types using the Morisita-Horn dissimilarity index calculated from species relative abundance data. Species relative abundance data were treated with a square-root transformation. Pairwise comparisons between banks and natural wetland types were made by conducting individual PERMANOVA tests for each comparison.

Habitat	Site Type Comparison	F	R^2	P
Emergent	Overall	4.39	0.15	<0.0001
	Bank - Random Low	6.33	0.13	<0.0001
	Bank - Random High	5.17	0.11	<0.0001
	Bank - Reference	5.05	0.17	<0.0001
Wet Meadow	Overall	7.17	0.24	<0.0001
	Bank - Random Low	6.05	0.11	<0.0002
	Bank - Random High	4.36	0.11	<0.0001
	Bank - Reference	8.90	0.25	<0.0001

Table 3.5. Five species with the highest average relative cover in each site type, for both emergent and wet meadow communities.

	Emergent		Wet Meadow	
Site Type	Species	Cover	Species	Cover
Bank	*Typha angustifolia**	26.98%	*Phalaris arundinacea**	17.23%
	*Phragmites australis**	13.18%	*Solidago canadensis*	12.88%
	*Phalaris arundinacea**	8.92%	*Silphium perfoliatum*	6.97%
	Carex lacustris	6.01%	*Zizia aurea*	5.01%
	Sparganium eurycarpum	5.50%	*Monarda fistulosa*	3.94%
Random Low	*Phalaris arundinacea**	43.59%	*Phalaris arundinacea**	46.79%
	*Typha angustifolia**	7.23%	*Solidago canadensis*	4.49%
	*Phragmites australis**	5.66%	*Persicaria amphibium*	3.27%
	Bolboschoenus fluviatilis	4.39%	*Aster lanceolatus*	2.69%
	Typha latifolia	3.79%	*Ambrosia trifida*	2.17%
Random High	*Phalaris arundinacea**	23.15%	*Carex stricta*	14.02%
	Bolboschoenus fluviatilis	6.03%	*Leersia oryzoides*	6.79%
	*Typha angustifolia**	4.82%	*Phalaris arundinacea**	6.58%
	Lemna minor	4.19%	*Calamagrostis canadensis*	5.12%
	Leersia oryzoides	4.03%	*Solidago canadensis*	4.32%
Reference	*Lemna minor*	15.60%	*Carex stricta*	26.42%
	Sagittaria latifolia	11.45%	*Calamagrostis canadensis*	12.78%
	Bolboschoenus fluviatilis	6.97%	*Thelypteris palustris* var. *pubescens*	6.41%
	Acorus americanus	5.45%	*Eupatoriadelphus maculatus*	6.37%
	Sparganium eurycarpum	3.76%	*Solidago gigantea*	4.17%

* Non-native species

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CHAPTER 4: EVALUATING THE ABILITY OF WETLAND MITIGATION BANKS IN THE CHICAGO REGION TO REPLACE NATIVE PLANT SPECIES LOST TO IMPACTS TO NATURAL WETLANDS

4.1 Introduction

The United States federal government practices a wetland mitigation policy that uses wetland offsetting to ensure that the country's wetland resources are not diminished by land development. Under Section 404 of the Clean Water Act of 1972, the US Army Corps of Engineers (Corps) and the US Environmental Protection Agency (EPA) regulate any land development projects which negatively affect areas legally delineated as wetlands. The Corps and EPA may allow public or private permittees to construct projects, which cause adverse impacts to existing wetlands, but permittees are then responsible to produce new wetland resources to offset these losses to natural wetlands. Permittees generally provide wetland compensation by constructing or funding projects that restore, create, enhance, or preserve wetlands (IWR 2015). These projects generate an increase in wetland area, function, and resources that is designed to offset the permitted wetland losses, so that there is "no overall net loss of [wetland] values and functions" (Corps and EPA 1990).

Permittees may choose to independently construct a wetland restoration project that satisfies their wetland compensation requirements, but the Corps has stated a preference that permittees meet these requirements through wetland mitigation banking (Corps and EPA 2008). When wetland mitigation banking is used, these wetland resources are produced in a large wetland restoration project constructed by a third-party bank sponsor. Mitigation banks are initially proposed by bank sponsors and permitted with a local Interagency Review Team (IRT), which is made up of representatives from the Corps, the EPA, and other natural resources agencies (Corps and EPA 2008). Once a bank project has been approved and permitted, the bank sponsor completes construction of new wetlands, restores previously existing wetlands, or enhances degraded wetlands. After a five-year management and monitoring period, the IRT evaluates a bank's compliance with ecological, primarily vegetation-based performance standards designed to ensure that the bank has created wetland resources that are equivalent to those in natural wetlands (IRT 2008). If the bank meets these standards then the IRT will release mitigation credits from the bank, and the bank sponsor may sell these credits to mitigation permittees, who use them to satisfy their mitigation requirements. The number of credits released from a bank is based on the type and acreage of wetlands or upland buffers produced in the bank, but generally one credit is released for every one acre of wetland that is restored (Corps and EPA 2008).

Many studies have used vegetation-based metrics and measures of ecological integrity to compare the plant communities in mitigation wetlands of all types, and specifically in mitigation banks, to those in natural wetlands. The results vary depending on the metrics used and the age of the mitigation projects being evaluated. Some have found that plant species richness in mitigation wetlands may approach that of natural wetlands (Gutrich et al. 2009, Matthews et al. 2009, Hopple and Craft 2013), but there is also evidence that species richness in natural sites exceeds that in mitigation wetlands (Campbell et al. 2002, Gutrich et al. 2009, Wall and Stevens 2015). Studies of mitigation banks, specifically, have produced mixed results. Some have found that mitigation bank wetlands and natural wetlands have similar plant communities (Spieles et al. 2006, Stefanik and Mitsch 2012), whereas others have reported that banks possess poor wetland function and ecological integrity relative to natural wetlands (Mack and Micacchion 2006, Reiss et al. 2007).

Previous studies of wetland mitigation have often used vegetation-based metrics to compare mitigation projects to natural reference wetlands (Brinson and Rheinhardt 1996, Fennessy et al. 2013), rather than seeking to determine if the plant communities in mitigation wetlands possess the specific plant species that were present in the natural wetlands they were intended to replace. I attempted to address this question of plant species replacement by combining a novel simulation modeling approach with biological data to evaluate the ability of wetland mitigation banks to replace the specific native plant species lost from natural wetlands during the mitigation process. I then used the results to assess which wetland mitigation policies may maximize the replacement of plant species lost in natural wetlands due to permitted impacts. The objectives of this study are to: 1) Determine the ability of wetland mitigation banks to replace the native plant species present in the natural wetlands for which they are used as compensation under typical regulatory conditions, 2) assess how species replacement varies by species floristic quality and for individual species, and 3) compare banks' ability to replace the native plant species in natural wetlands under different regulatory conditions governing the credit transactions between natural wetlands and banks.

4.2 Methods

To evaluate the ability of wetland mitigation banks to replace the native plant species found in impacted natural wetlands, I used plant species lists from two sources in northeastern Illinois, USA: 1) wetland mitigation banks and 2) natural wetlands that are characteristic of the type of wetlands likely to be impacted and compensated for under wetland mitigation policy.

Wetland mitigation banks

I conducted this study using wetland mitigation banks that were permitted and regulated within the Chicago District of the Corps. The Chicago District includes six counties in northeastern Illinois: McHenry, Lake, Cook, Kane, DuPage, and Will. This regulatory district is an ideal region in which to study wetland mitigation banking because it possesses an extensive and well-developed wetland banking market that includes some of the oldest private wetland banks in the country. I used data which were collected by bank sponsors to satisfy their regulatory requirements. Bank sponsors are typically required to perform annual management and monitoring of banks for at least five years following their construction, after which the IRT may grant final credit release if the bank has met its ecological performance standards (Corps and EPA 2008). The results of this monitoring are summarized and submitted to the IRT in an annual monitoring report which includes general project information, summaries of credit releases granted by the IRT, and site-wide plant species lists. I obtained these monitoring reports, as well as information about the number of credits generated in each bank, for all banks that have been regulated by the Chicago District IRT. I acquired this information from the Corps' online database RIBITS (Regulatory In-lieu Fee and Bank Information Tracking System) (Corps 2020), the US EPA Region 5 Main Office in Chicago, and from bank sponsors.

There are more than 20 wetland mitigation banks that have been regulated by the IRT in the Chicago District, but I was only able to include 13 of these in my analysis. I excluded banks that did not achieve final approval and credit release from the IRT, those for which I could not find comprehensive plant species lists from the appropriate monitoring period, and those for which I could not accurately determine the total number of credits that had been released. One of the banks in my study was constructed in several non-adjacent phases that were permitted and administered as separate projects by the IRT; I included two of these as separate sites in my analysis. Each of the 13 banks I selected was constructed between 1997 and 2007 and had received its final credit release from the IRT by the end of 2017. Banks ranged in size from 24.9 acres to 94.0 acres (mean = 59.7 acres), producing between 18.4 and 80.3 credits (mean = 48.3).

I obtained sitewide plant species lists from annual monitoring reports for each bank, in every year for which I could locate a monitoring report. The data collection protocols that bank sponsors used to produce these species lists varied between banks and between monitoring years. The monitoring reports typically included a full site species list, as well as species lists produced during required quadrat monitoring and during more localized species searches. Often the lists reported were the result of multiple sampling efforts made throughout the growing season, but the number of sampling visits was not consistent across banks. My correspondence with bank sponsors suggested that the full site species list given in most monitoring reports usually included every species that the bank sponsor observed from the entire site throughout the year. I took this same approach in compiling sitewide species lists from the monitoring reports and included in my lists every species that was reported from a bank across all sampling scales and all sampling visits.

I chose to use only one year of monitoring data from each bank and limited my selection to data that were collected from banks that were between four and six years old at the time of monitoring. This range approximates the five-year age at which the IRT expects to evaluate and give final site approval to banks. The typical monitoring and credit release schedule was followed for many banks, but some received final approval as early as their second year, while others were required to perform annual management and monitoring for up to 12 years before the IRT released all credits from the bank. I used the data from a bank’s fifth monitoring year if possible, but then accepted data from banks’ fourth or sixth monitoring years if fifth year data were not available.

To simulate the purchase of credits from banks I had to determine how many credits were released from each bank. During the initial permitting of a bank, the IRT and the bank sponsor agree upon the amount of credits the bank is expected to generate; however, the final number of credits released may differ from this initial estimate if the bank fails to produce the expected acreage of wetlands meeting performance standards. A credit ledger detailing the credit releases for each bank is available from RIBITS, but this information is incomplete or inaccurate for some banks. When possible, I supplemented the RIBITS ledger with credit release records found in the monitoring reports to determine the total number of credits that had been released from each bank at the time of final credit release.

Natural wetlands

I obtained data from a dataset of natural wetlands that initially included 2,005 sites within the geographic boundaries of the Chicago District of the Corps. Data from specific natural wetlands that were impacted under wetland mitigation policy are not available; however, the natural wetlands in my study were all surveyed as part of planned road construction and maintenance projects for the Illinois Department of Transportation (IDOT), so they represent the types of natural wetlands that are likely to be impacted under wetland mitigation policy (Skultety 2015). Data including geographic coordinates, wetland plant community type, the delineated wetland area, and a comprehensive plant species list resulting from a species search of the entire wetland were collected from these sites by the Illinois Natural History Survey (INHS) between 2002 and 2013. INHS botanists conducted all surveys between April and October in each sampling year. Within the Chicago District, wetland mitigation has, in most cases, only been required when natural wetlands are affected by negative impacts that exceed 0.1 acres (US Army Corps of Engineers Chicago District 2012). I applied this limit to my natural wetland data and excluded sites that measured less than 0.1 acres, which left 1,530 remaining natural wetlands sites ranging in size from 0.1 acres to 193.1 acres.

Plant data preparation

Plant species lists from banks and natural wetlands were originally recorded using different authorities for plant nomenclature. To allow direct comparison between these data, I converted all species names from both banks and natural wetlands into the nomenclature used by Swink and Wilhelm (1994) in their regional flora of the Chicago region. This is the naming authority that was used by most bank sponsors, and under which the ecological performance standards used for the banks in my study were developed. I excluded from my analysis plants that were identified only to genus because I could not determine for these records if direct species replacement occurred. I also excluded all non-native species from my analysis because the replacement of non-native species by mitigation banks is not a goal of mitigation policy, nor is it desirable.

For each plant species occurring in banks or natural wetlands, I obtained coefficients of conservatism (C-values) from Swink and Wilhelm's 1994 flora. The C-value is a tool developed by Swink and Wilhelm that is used to quantify and assess the floristic quality of plant species and communities. Swink and Wilhelm assigned to each native plant species in the region a C-value ranging from 0 to 10 that measures the species' fidelity to natural habitats. Species were given low scores if they were known to be tolerant of habitats experiencing a high degree of anthropogenic disturbance, whereas species were given high scores if they were known to be conservative, occurring only in undegraded natural communities.

Simulation model

To evaluate the species replacement achieved by banks, I created a model to simulate the destruction of natural wetlands and the resulting exchange of credits that occurs when wetland mitigation permittees are required to purchase mitigation credits from a bank. I compared the plant species occurring in natural wetlands impacted in these simulations to those in mitigation banks so that I could assess species replacement. I created this model using R 3.3.0 software (R Core Team 2020). I included each mitigation bank and natural wetland as a separate site in the model. The model simulated the mitigation process using one mitigation bank at a time. During a single run of the model, one natural wetland at a time was randomly selected to be "destroyed," the appropriate number of credits required to be purchased was determined based on the area of the destroyed natural wetland, and these credits were withdrawn from the total number of credits produced by the bank. The model continued to select and destroy natural wetlands until at least 90% of the credits available in the bank had been purchased. Terminating each model run after reaching the 90% threshold was necessary for the model to function, as it could not selectively choose small credit purchases to sell every credit for each bank nor did I allow it to complete partial credit purchases to sell the remaining credits; however, this means that banks in my model did not sell all their credits, and that my model likely underestimates the number and acreage of natural wetlands that would have used these banks as compensation. After terminating credit purchases, the model retrieved the list of native plant species from each natural wetland that had been destroyed during the model run and compiled these into a list of all the species present in at least one of the impacted natural wetlands. The model also retrieved the species list from the single wetland mitigation bank used during the model run. Each species in these lists was then grouped into one of three categories: (1) *species replaced* were present in both the destroyed natural wetlands and the bank, (2) *species lost* were present in the destroyed natural wetlands but were not present in the bank, and (3) *species gained* did not occur in any of the destroyed natural wetlands but were present in the bank. I calculated the percentage of species replaced for the model run by dividing the number of species replaced by banks by the total number of species present in the destroyed natural wetlands. I repeated this simulation 1,000 times for each mitigation bank to obtain average values for species replacement and the number of species replaced, lost, and gained for each bank. I interpreted the results of these average values across all banks.

Certain parameters in the model represent real-world policy conditions that regulate credit transactions. The number of mitigation credits required to compensate for permitted wetland losses is determined by the Corps based on a mitigation ratio, which is the ratio between the acres lost in natural wetlands and the number of mitigation credits required. At the federal level, the Corps sometimes requires a minimum 1:1 mitigation ratio, but in many cases this minimum ratio may be increased, requiring the permittee to purchase more mitigation credits than the number of acres they have impacted (Corps and EPA 2008). The Corps may change the mitigation ratio depending on the likelihood that the mitigation project will be successful, differences between the wetland functions or resources impacted and those that will be produced via mitigation, and the distance between the impact and mitigation site (Corps and EPA 2008). In the Chicago District, the minimum mitigation ratio applied is typically 1.5:1 (US Army Corps of Engineers Chicago District 2009). In my model, I was able to manipulate the mitigation ratio used to determine how many credits needed to be purchased from the bank for each natural wetland. I ran the base model using a mitigation ratio of 1.5:1, following the default used in the Chicago District.

To address my second research objective, I used the base model to calculate species replacement figures using all native species, but I also calculated replacement for species grouped by C-value into categories of low (0-2), medium (3-7), and high (8-10) floristic quality. This allowed me to assess if banks' ability to replace the species lost in natural wetlands is different for highly conservative species than it is for disturbance-tolerant species.

I also examined species replacement outcomes for individual plant species by conducting a run of the base model in which I calculated the percentage of trials across all banks in which each plant species was replaced, lost, gained, or absent from the simulation. I sorted these by the simulation outcome, and reported the 25 species most frequently replaced, lost, and gained.

Testing different policy conditions

To test the effect that changes to certain policy conditions would have on species replacement, I varied the model parameters that represent these policies.

When possible, the Corps prefers that wetland compensation be "in-kind," so that mitigation wetlands are of the same structural and functional type as the natural wetlands for which they are used as compensation (Corps and EPA 2008). The wetland banks in my study produced exclusively open herbaceous wetlands, though some banks may have had small, forested inclusions. The natural wetlands I assessed were classified by INHS botanists into several different plant community types, including open herbaceous wetlands (846 sites after removing wetlands measuring less than 0.1 acres), forested wetlands (209 sites), and shrub/scrub wetlands (68 sites); additionally, 407 sites were not given a plant community classification. In my model, an "in-kind only" approach would be achieved by including only open herbaceous natural wetland sites, so that mitigation banks could only be used as compensation for natural wetlands of the same plant community type as was produced in banks. I do not know whether the

Chicago District IRT has historically followed an in-kind only approach when approving credit sales from banks, but I choose to use this as the default approach for my model, so that I only included the 846 natural wetlands classified by INHS botanists as open herbaceous plant communities when running the base model and for models run to test all other policy conditions. However, to compare the species replacement that could be achieved when using in-kind only compared to non-restricted mitigation, I also conducted a model run in which I included natural wetland sites of all plant community types, including those for which a plant community was not recorded. This model run included all 1,530 natural wetland sites that were larger than 0.1 acres. I conducted 1,000 simulations for both the in-kind only and the non-restricted model runs.

To test the effect that simply increasing the mitigation ratio would have on species replacement, I compared the results of simulations run using mitigation ratios of 1.5:1, 3:1, and 6:1, conducting 1,000 simulations for each ratio.

I also conducted a simulation in which the mitigation ratio for each natural wetland site was scaled based on the floristic quality of that site. I modeled this approach after a wetland mitigation policy that is used in collaboration with Army Corps regulations by a county government within the Chicago District of the Corps (Kane County 2019). This policy uses a measure of sitewide floristic quality called the native Floristic Quality Index (FQI), which is calculated using the total number of native species present at a site (S) and the average of the C-values from all the native species present in the site (Mean C) (Swink and Wilhelm 1994):

$$\text{FQI} = \text{Mean C} \times \sqrt{\text{S}}$$

The Kane County policy increases the mitigation ratio for natural wetlands when they pass several native FQI thresholds. I made minor simplifications to this approach but used the same thresholds and ratios in my model. Natural wetlands with a native FQI less than 7 were mitigated at a 1:1 ratio, natural wetlands with a native FQI between 7 and 16 were mitigated at a 2:1 ratio, natural wetlands with a native FQI between 16 and 25 were mitigated at a 3:1 ratio, and natural wetlands with a native FQI greater than 25 were mitigated at a 10:1 ratio. I compared the results of this simulation to those of the base model.

The IRT restricts the geographic area within which wetland mitigation banks can sell credits; this is referred to as the bank's service area. The boundaries of service areas can be selected based on natural features such as watershed, ecoregion, or physiographic boundaries (Corps and EPA 2008) but they may also be defined by political boundaries such as counties, entire states, Corps districts, and Department of Transportation Districts (Brown and Lant 1999). In some cases, local governments may institute their own mitigation requirements that affect bank service area. For example, while the Chicago District allows mitigation banks to sell credits within one service area that covers nearly the entire District (IRT 2017), some counties within the District require that wetland mitigation be provided within the county where wetland impacts occur (Lake County 2015, Kane County 2019). To test the effect these county-level policies have on species replacement, I designed my model with an option to restrict credit transactions so that they could only occur between banks and natural wetlands that were in the same county. One of the counties in the Chicago District contained just one of the mitigation banks included in my study, but did not encompass enough natural wetland sites to exhaust all of the credits present in that bank; therefore, I excluded that bank from this analysis and made this comparison using only 12 banks that were distributed across three different counties. I ran this comparison using the 1.5:1 mitigation ratio used for the base model and compared the results of the county-restricted model to those of the base model with no county restriction.

Numerical analysis

For the base model analysis, I calculated the sample mean and standard deviation for the average number of species replaced, lost, and gained, and the percentage of species that were replaced, for each trial. I also used a t-test to obtain the estimated mean and 95% confidence interval for average species replacement under the base model.

I used linear mixed modeling to test for differences in species replacement between different policy conditions. I constructed four different models to test the four policy approaches (in-kind only vs non-restricted mitigation, increasing mitigation ratio, using a mitigation ratio scaled by native FQI, and requiring within-county mitigation). In each case I used the policy condition as a fixed effect, with treatment levels that included the selected policy change and the base model. Re-running the model under different policy conditions collects multiple samples from each mitigation bank, so I included bank as a random effect in all linear mixed model comparisons. I had no a priori reason to believe that the effect of policy conditions would vary for different banks, so I conducted all testing using random intercepts models. After fitting each model, I examined residual plots to ensure that the data met model assumptions of homoskedasticity and normality for residuals; these assumptions were valid in all cases. For each policy comparison, I used model outputs to obtain parameter estimates and 95% confidence intervals for the mean percent species replacement for each policy level. I also obtained marginal R^2 values (representing the variance explained by the fixed effect alone) and conditional R^2 values (representing the variance explained by the fixed effect and the random bank effect) using the r.squaredGLMM function from the MuMIn package in R (Barton 2020). To conduct a hypothesis test for each policy model, I used likelihood ratio tests to determine if adding the policy fixed effect improved model fit over the model run with only the random effect. My only policy condition with more than two treatment levels was mitigation ratio; I did not attempt to make pairwise comparisons between these treatment levels.

4.3 Results

Base model and species comparisons

When I ran the base model, with a mitigation ratio of 1.5:1, banks were used to compensate for an average of 29.6 natural wetland sites and 29.9 natural wetland acres. To illustrate the difference between within-bank and between-bank variation in species replacement, the distribution of species replacement results for all 1,000 trials at each bank are given in Figure 4.1. Figure 4.2 shows the distribution of mean percent species replacement values for all native species across all 13 banks, averaged over 1,000 trials for each bank. Based on sample means calculated from the base model (Table 4.1), the greatest number of species are lost in mitigation transactions (mean = 85.2), followed by replaced species (mean = 68.4), and then species gained by banks (mean = 48.2). The parameter estimate for percent species replacement obtained from a t-test was 45.2% with a 95% confidence interval of 39.0% to 51.4%, which matches the sample mean exactly (Table 4.1).

Comparing between floristic quality groups (Figure 4.2, Table 4.1), percent species replacement was highest for the low C group (sample mean = 50.4%, *SD* = 16.1%), though there did not seem to be a large or significant difference between the low C and medium C groups; however, species replacement among high C-value species was much lower (sample mean = 12.3%, *SD* = 8.1%).

Results are also presented for the average number of species replaced, lost, and gained (Figure 4.3, Table 4.1). Across all native species, an average of 48.2 species (*SD* = 21.6) were gained per bank. The average number of species gained was highest for plants with medium C-values and was much lower for both low C and high C species. Overall, fewer high C species were involved in the model (replaced, lost, or gained) than from either of the other C-value groups.

The percentage of trials in which individual species were replaced, lost, gained, or absent from both destroyed natural wetlands and banks is presented for the species most frequently replaced (Table 4.2), most frequently lost (Table 4.3), and most frequently gained (Table 4.4).

Comparisons of different policy conditions

The likelihood ratio test for a difference in mean percent species replacement between in-kind only and non-restricted mitigation indicated that mean species replacement at a single bank was significantly greater ($\chi^2 = 42.5$, $P < 0.001$, marginal $R^2 = 0.02$, conditional $R^2 = 0.99$) when allowing only in-kind mitigation (45.2%) than for non-restricted mitigation (42.0%) (Figure 4.4, Table 4.5).

There was a significant overall difference in mean species replacement among different mitigation ratios ($\chi^2 = 62.6$, $P < 0.001$, marginal $R^2 = 0.07$, conditional $R^2 = 0.99$), which resulted in estimates for percent species replacement that moderately increased for higher ratios: 45.1% at 1.5:1, 49.1% at 3:1, and 52.3% at 6:1 (Figure 4.5, Table 4.5). Increases in mitigation ratios should produce proportionate decreases in the average amount of natural wetland acres that may be compensated for by a single bank (i.e. doubling the mitigation ratio should result in banks being used to compensate for only half as many natural wetland acres). Examining the average number of acres impacted at different mitigation ratios under my model revealed that the model was successfully representing this consequence of changes to the mitigation ratio (Table 4.5).

Likelihood ratio testing for a difference in mean species replacement between the base model and the model run with a scaled mitigation ratio revealed a statistically significant difference ($\chi^2 = 28.8$, $P < 0.001$, marginal $R^2 = 0.01$, conditional $R^2 = 1.0$), but the small marginal R^2 value and the small difference between estimated species replacement (45.1% under the base model and 47.2% with a scaled mitigation ratio) indicated that this policy produced only a small improvement (Figure 4.6, Table 4.5). This policy functions by changing the mitigation ratio for high-quality natural wetland sites, but the resulting change in the number of acres impacted was only moderate, as the average acres impacted using the scaled ratio (21.6) was only 28% less than the average acres impacted under the base model (29.9) (Table 4.5).

Allowing only within-county credit transactions in the model produced no difference in percent species replacement compared to the base model ($\chi^2 = 0.7$, $P = 0.40$, marginal $R^2 < 0.001$, conditional $R^2 = 0.97$, Figure 4.7, Table 4.5).

4.4 Discussion

Overall species replacement

Under my base model, run using a mitigation ratio of 1.5:1 that represents the lowest default ratio that would generally be required by regulatory agencies in the Chicago District, banks replaced an average of 45% of the native plant species present in impacted natural wetlands. This result suggests that many of the native plant species present in natural wetlands that are vulnerable to development are simply not present in many of the mitigation banks used as compensation. Others have found that restored wetlands do not include the full suite of plant species present in natural wetlands (Seabloom and van der Valk 2003), and that plant species composition in restored wetlands is distinct from that in natural wetlands (Wall and Stevens 2015). Banks' ability to replace impacted species is influenced, in part, by their ability to generate wetlands with an adequate number of native plant species. The evidence is mixed as to whether species richness in restored wetlands typically exceeds that in natural wetlands (Gutrich et al. 2009, Matthews et al. 2009, Hopple and Craft 2013) or falls short of richness levels in natural wetlands (Campbell et al. 2002, Gutrich et al. 2009, Wall and Stevens 2015). Wetland mitigation banks, specifically, may produce measures of species richness that are similar to those in reference wetlands (Spieles et al. 2006). Another study found that species richness in banks compared poorly to that in natural wetlands, but that more recently constructed banks may compare favorably (Stefanik and Mitsch 2012).

Relationship between species characteristics and replacement

I found that species replacement was particularly low for highly conservative native species. This result indicates that banks did not effectively replace species with a high affinity to undegraded wetland communities, which are also likely to be among the least common species included in my dataset. There is evidence that restored wetlands typically lack regionally uncommon species; one study from the prairie pothole region found that 70% of the infrequent species and 93% of the rare species from natural wetlands in the region never occurred in restored wetlands (Aronson and Galatowitsch 2008). The loss of conservative species may be especially significant if their regional distribution in natural wetlands is limited, especially in a highly developed urban area such as the Chicago metropolitan region.

I can make a number of observations about the outcome of wetland mitigation in banks by examining the species that were most frequently replaced, lost, and gained during simulations of wetland mitigation banking, taking into account the characteristics and habitat preferences of these species described in Swink and Wilhelm's 1994 flora. The species most frequently replaced (Table 4.2) were made up mostly of forbs and graminoids, including some of the most common wetland species in the region. Most of these species are common in a variety of wet habitats, including marshy ground and moist meadows (e.g. *Verbena hastata*, *Scirpus atrovirens*, *Juncus dudleyi*, and *Asclepias incarnata*). This list also includes several annual species that are common in weedy and early successional habitats, such as cultivated fields and waste areas (e.g. *Ambrosia trifida*, *Echinochloa crusgalli*, and *Ambrosia artemisiifolia* var. *elatior*). Most of the species in this list have moderate to low C-values, indicating a tolerance for a variety of natural, and sometimes degraded, habitats. Very high replacement percentages among species at the top of this list indicate that they were present in every bank in my study and within at least one natural wetland in almost every model run. Overall, the species most often replaced can be characterized as abundant wetland habitat generalists.

The species most frequently lost during model simulations (Table 4.3) included several woody plants and herbaceous species that are often found in wooded habitats. These included shrubs characteristic of different habitat types, including woodlands and edge habitats (*Sambucus canadensis*), more open areas along streams or ponds (*Salix amygdaloides* and *Cornus obliqua*), and moist woods (*Ribes americanum*). There were also several forb and vine species that are most often characteristic of wooded habitats, though sometimes occurring in open areas as well (e.g. *Parthenocissus quinquefolia*, *Impatiens capensis*, *Rhus radicans*, and *Galium aparine*). Species commonly lost also included canopy trees characteristic of rich mesic woods and floodplains (*Ulmus americana* and *Fraxinus pennsylvanica* var. *subintegerrima*) and fully aquatic species (*Ceratophyllum demersum* and *Potamogeton nodosus*). In the base model used for this analysis, I excluded forested natural wetlands and included only natural wetland sites classified as belonging to open herbaceous plant community types, but these woody plants and species characteristic of forested floodplains and wet woods were present in the sites I did include, nonetheless. Their presence on this list suggests that wetland mitigation banks did not often include these species, even though they were represented among natural wetlands that were permitted to be impacted under in-kind only mitigation. All the wetland mitigation banks included in my study restored open wetland community types dominated by herbaceous species (wet prairies, sedge meadows, and marshes), as well as areas of mesic prairie buffer; however, my review of bank sites did show that areas of floodplain forest were present in a few banks. This may explain why most of the woody species that were commonly lost *were* successfully replaced in a small, but significant number of simulations. These forested areas in banks were usually pre-existing communities that did not receive additional restoration and were not permitted to produce wetland credits, but their species would likely have been included on the site species lists used in this study.

The species most frequently gained (Table 4.4) are composed primarily of perennial sedges, forbs, and grasses. If these species were not often present in natural wetlands, and were frequently gained in banks, then their presence in banks was likely due to seeding or planting, which were conducted at all banks to restore desirable plant communities. It is noteworthy that the species most often gained in banks tended to have relatively high to moderate C-values. To receive their final credit release from the IRT, the banks in my study were required to exceed a minimum value for the average of all C-values from native species at the bank (IRT 2008). This performance standard created an incentive for bank sponsors to establish species with high C-values in banks, which offers a possible explanation for their presence on the list of species most frequently gained. Many of these species are commonly included in seed mixes for prairie restorations in the region, including species that are characteristic of wet prairies and sedge meadows (*Carex scoparia*, *Physostegia virginiana*, *Solidago riddellii*) and others characteristic of dry to mesic prairies (*Elymus canadensis*, *Ratibida pinnata*, *Andropogon gerardii*, *Sorghastrum nutans*). Banks typically included areas of upland prairie buffer which were able to generate wetland credits, though at a lower rate than areas of restored jurisdictional wetlands (IRT 2008). The dry and mesic prairie species that were often gained by banks may represent these buffer communities, which would not have been present in the natural wetlands included in my model.

If banks have been unable to replace many of the plant species found in impacted natural wetlands, as my data indicate, then targeting seeding and planting efforts to replace "lost" species could be a logical approach to improve species replacement. Most of the banks in my study were converted from areas of commercial agricultural production. Depending on the site-specific land use history and intensity of farming practices, this could leave most bank sites with a seedbank that lacks many of the region's native wetland species. Indeed, overall species richness, wetland species richness, and the richness of certain key wetland taxa has been found to be lower in the seedbanks of restored wetlands than in adjacent natural wetlands (Wall and Stevens 2015). Increasing the number of species that are seeded or planted may increase the similarity of plant species composition between mitigation wetlands and natural wetlands (Matthews and Spyreas 2010), though it may not always increase native species richness and floristic quality (Matthews and Endress 2008).

Changes to policy conditions

I found that requiring in-kind only mitigation in my model did improve species replacement in banks; however, this improvement was modest, as mean species replacement using in-kind only mitigation was only 3.2% greater than non-restricted mitigation, and the model for this comparison had a low goodness of fit (marginal R^2 value) of 0.02. While in-kind only mitigation did improve species replacement, the banks in my study did not produce forested wetlands, and so would not have been able to provide in-kind mitigation for impacts to forested natural wetlands. Forested wetland restoration presents a difficulty for wetland mitigation policy because these projects require a greater length of time to complete and evaluate than do restorations of wetland types dominated by herbaceous plants. When the banks in my study were active, the Chicago District IRT did not seem to have an official regulatory framework with which to evaluate forested wetlands in mitigation banks, as the performance standards used by the IRT at this time were written only for herbaceous wetlands and mesic prairie buffers (IRT 1997). The performance standards established most recently by the IRT *can* accommodate floodplain forests (IRT 2017), which may encourage the restoration of this habitat type in future bank projects, increasing the availability of credits that could be used to satisfy demand for in-kind mitigation of forested wetlands.

The Army Corps may increase the mitigation ratio with the goal of improving mitigation success in situations where high quality natural wetlands are affected by permitted impacts, mitigation projects are unlikely to succeed, or there is a difference between the wetland resources affected and those produced by mitigation (Corps and EPA 1990, 2008). Implementing a mitigation ratio scaled by native FQI is designed to accomplish a similar purpose, as more credits are required to compensate for natural wetlands with plant communities that possess greater floristic quality, presumably because the task of compensating for these wetland resources is less likely to succeed. While I found that these two policy conditions both resulted in a statistically significant increase in percent species replacement, the mixed linear models for these comparisons had low goodness of fit (marginal R^2 values) and the actual increase in percent replacement was only moderate for increasing the mitigation ratio and was very low for using a ratio scaled by native FQI. Increasing the mitigation ratio does provide some opportunity for improving the replacement of the specific wetland resources found in impacted wetlands, but there may be a limit to the gain in species replacement that can be achieved with these policies.

While increasing mitigation ratios is sometimes used to improve wetland compensation when high-quality natural wetlands will be affected, it is important to note that there are other protections in place to prevent the loss of high-quality wetland resources. Advanced Identification (ADID) studies have been conducted in counties within the Chicago District to identify high-quality wetland areas that are considered “generally unsuitable” for permitted projects that will cause adverse impacts (Dreher et al. 1992, Northeastern Illinois Planning Commission et al. 1998). Therefore, the Army Corps may protect high-quality natural wetlands in the Chicago District by declining to grant permits for potential impacts to these sites, rather than relying on wetland mitigation banks to compensate for these high-quality wetland resources.

I found that restricting wetland mitigation transactions to sites occurring within the same county as banks did not cause a change in percent species replacement. The Army Corps’ 2008 Mitigation Rule states that service areas should be established to “ensure that the aquatic resources provided will effectively compensate for adverse environmental impacts across the entire service area” (Corps and EPA 2008). Considering only the replacement of specific plant species, it seems that banks would compensate for species losses just as effectively with a service area covering the whole Chicago District as they would if restricted to county-level service areas. It is possible that service areas designed to follow ecologically meaningful boundaries, such as watershed boundaries, may have a greater effect on the replacement of natural wetland resources than those based on political boundaries. Of course, my results do not account for the effect of service area on banks’ ability to compensate for other structural and functional wetland resources, such as water storage, sediment retention, and wildlife habitat. Assessing the effectiveness of service area restrictions also requires an understanding that wetland mitigation banking leads to the spatial redistribution of wetlands. Between 1994 and 2002, wetland mitigation banking caused a net gain in wetland acreage across the entire Chicago District, but at finer spatial scales some watersheds experienced a net loss of wetland area (Robertson and Hayden 2008). Wetland mitigation banking tends to shift wetlands in a downstream direction (Brown and Lant 1999) and from areas of higher to lower population density (Ruhl and Salzman 2006). While plant species replacement was not benefitted by prohibiting credit transactions across county lines, county and city governments do have incentives to implement mitigation policies ensuring that other wetland resources will not be lost from their jurisdictional boundaries. The Army Corps, however, does have to ensure that bank service areas are large enough to provide sufficient credit demand to support banks (Corps and EPA 2008).

Conclusion

One of the stated goals of wetland mitigation policy is to ensure that there is "no *overall net* [emphasis added] loss of values and functions" of wetlands; however, this does not guarantee that *every* wetland resource will be preserved in all mitigation transactions (Corps and EPA 1990). This is evident in the way wetland banking differs from on-site mitigation, which requires that wetland compensation be provided at the same site at which permitted wetland impacts occur. Wetland banks redistribute and consolidate wetlands by compensating for multiple spatially distributed wetland impacts at one mitigation site (Brown and Lant 1999). Rather than requiring that all wetland impacts be offset by on-site compensation, which would constitute no *absolute* loss of wetlands, mitigation banking allows for compensation to be redistributed geographically while ensuring that the *net* quantity of wetland resources is maintained (Brown and Lant 1999). Emphasis on the overall net conservation of wetlands also has implications for which types of wetland resources are prioritized for compensation. The ecological performance standards banks are required to meet are usually based on structural wetland resources, most often with a focus on characteristics of the vegetation in banks (Environmental Law Institute 2002, Matthews and Endress 2008, Reiss et al. 2009). These standards may be designed to ensure that banks produce wetlands of equivalent community types to natural wetlands in the region, but they are measured by general metrics such as species richness and native dominance (IRT 2017) rather than requirements that banks produce the specific wetland components (e.g. the replacement of specific plant species) found in the natural wetlands for which they compensate. The Army Corps and EPA, while certainly concerned with conserving the specific structural components of wetlands, have more clearly identified wetland function and wetland area as the primary resources to be conserved through the mitigation process (Corps and EPA 1990).

For these reasons, replacement of the specific plant species found in impacted natural wetlands is not an explicit goal of wetland mitigation banking and the absolute loss of some species from impacted natural wetlands may be acceptable if no net loss of wetland resources is achieved. Therefore, it is difficult to objectively identify what should be an appropriate goal for plant species replacement. While an appropriate goal may be difficult to determine, what I have demonstrated with this project is an outcome of wetland mitigation banking: that many of the native plant species present in natural wetlands that may be impacted are simply not present in the wetland banks used as compensation. Even under a high mitigation ratio of 6:1, banks replaced, on average, just over half of the species in impacted sites. If the preservation of specific plant species and communities is to be a goal of wetland mitigation banking, banking practices and policies will need to achieve greater equivalence between the resources present in natural wetlands and those in banks. While replacement of wetland function and area may be the primary goal of wetland mitigation, it is necessary to understand what specific wetland resources and components may be lost in the mitigation process from banks that are deemed a regulatory and ecological success. Very little work has been done to determine how effectively wetland mitigation sites can replace specific wetland resources; this study has begun to provide this information and has introduced a novel approach for assessing the ecological outcome of wetland mitigation banking.

4.5 Figures

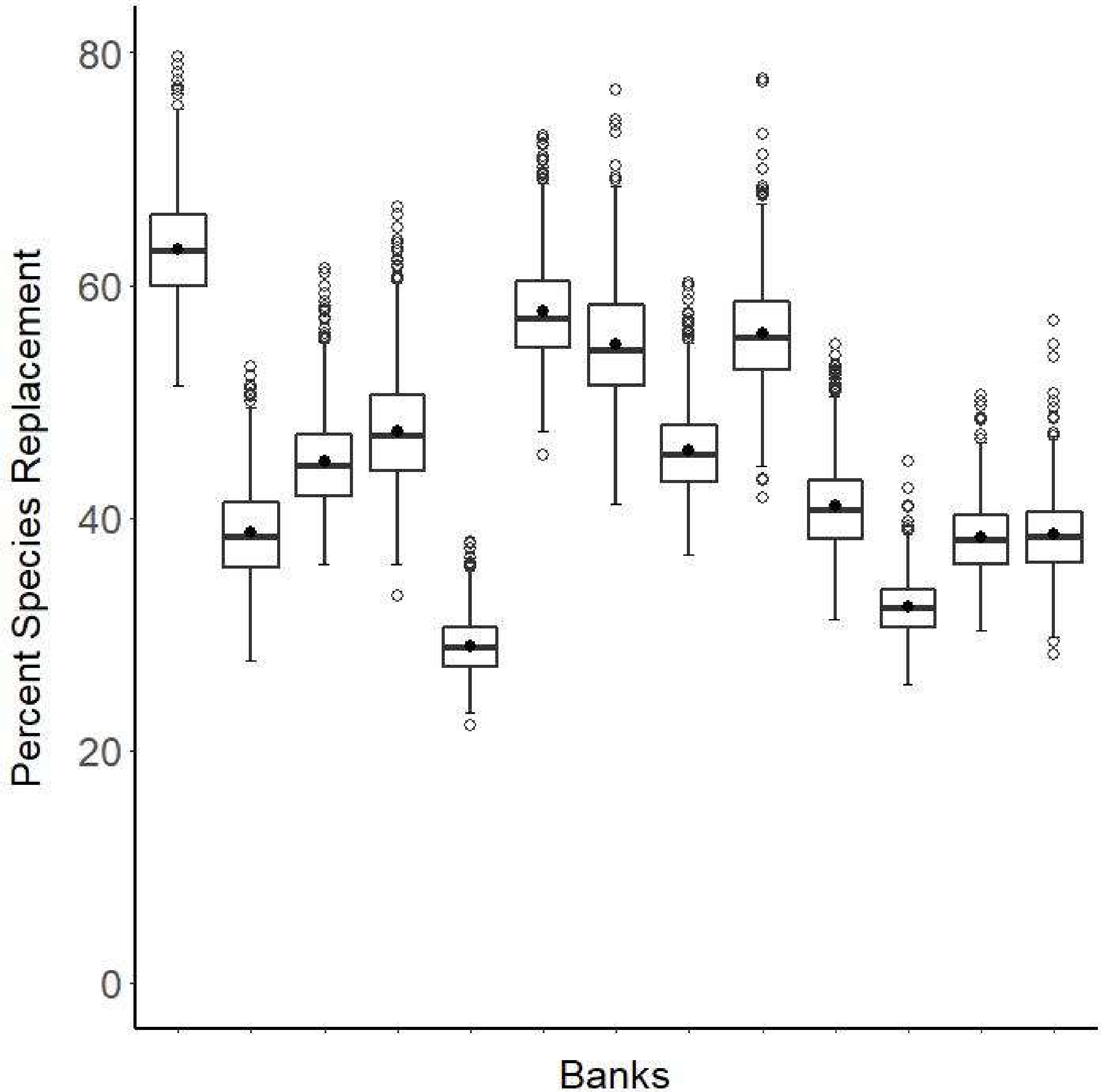

Figure 4.1. Box-and-whisker plots showing the distribution of simulation trial results for the percentage of native plant species that were replaced by wetland mitigation banks across 1,000 trials at each of 13 banks. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). All trials were run using a mitigation ratio of 1.5:1.

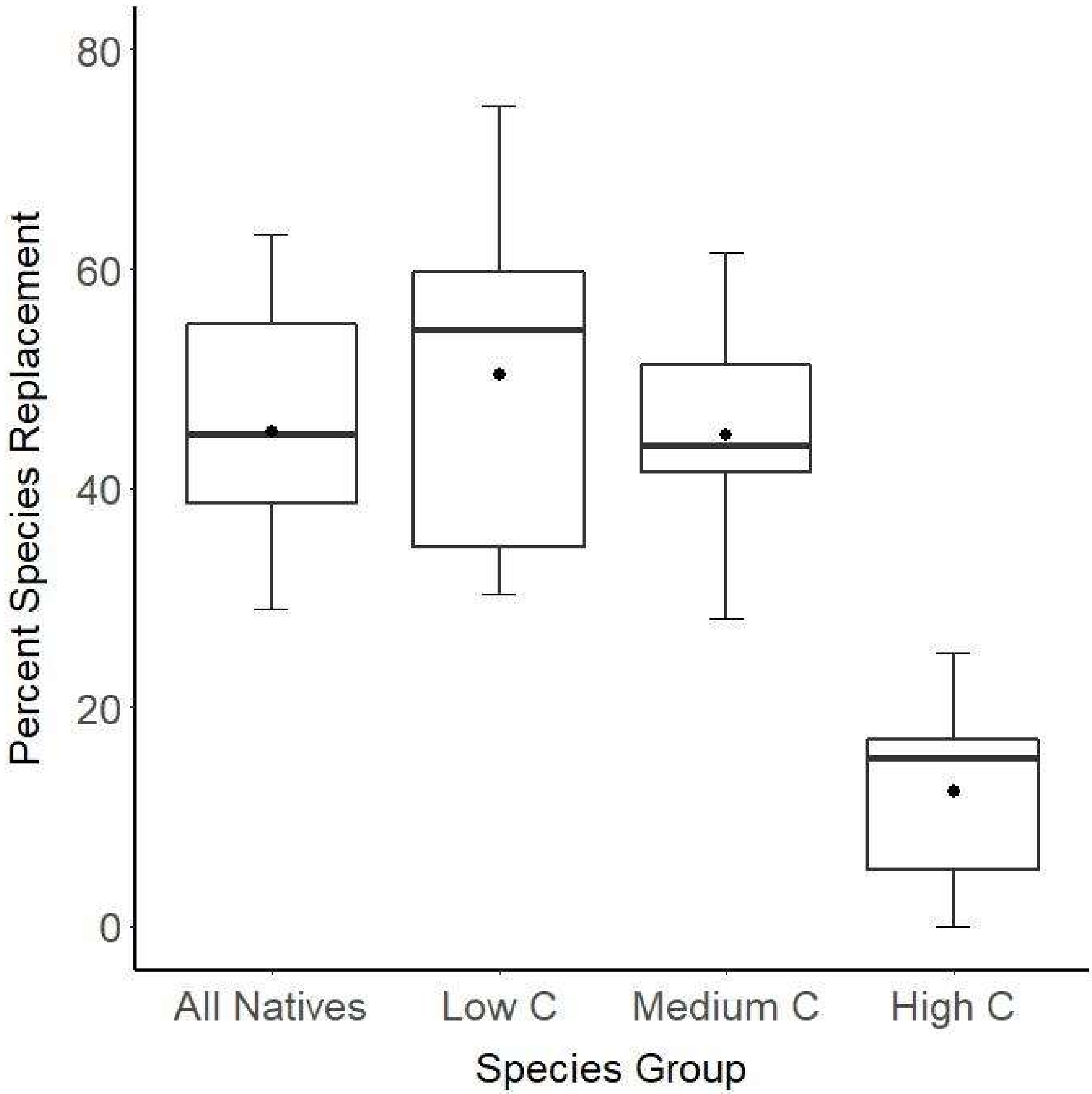

Figure 4.2. Box-and-whisker plots showing the average percentage of native plant species that were replaced at each wetland mitigation bank for all native species and across floristic quality groups of plant species. Box-and-whisker plots show the distribution of average replacement values from 13 banks, with the value for each bank being the average of 1,000 trials for that bank. The floristic quality groups contain the following range of C-values: 0-2 in Low C, 3-7 in Medium C, and 8-10 in High C. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), and the distribution of data points (whiskers). All trials were run using a mitigation ratio of 1.5:1.

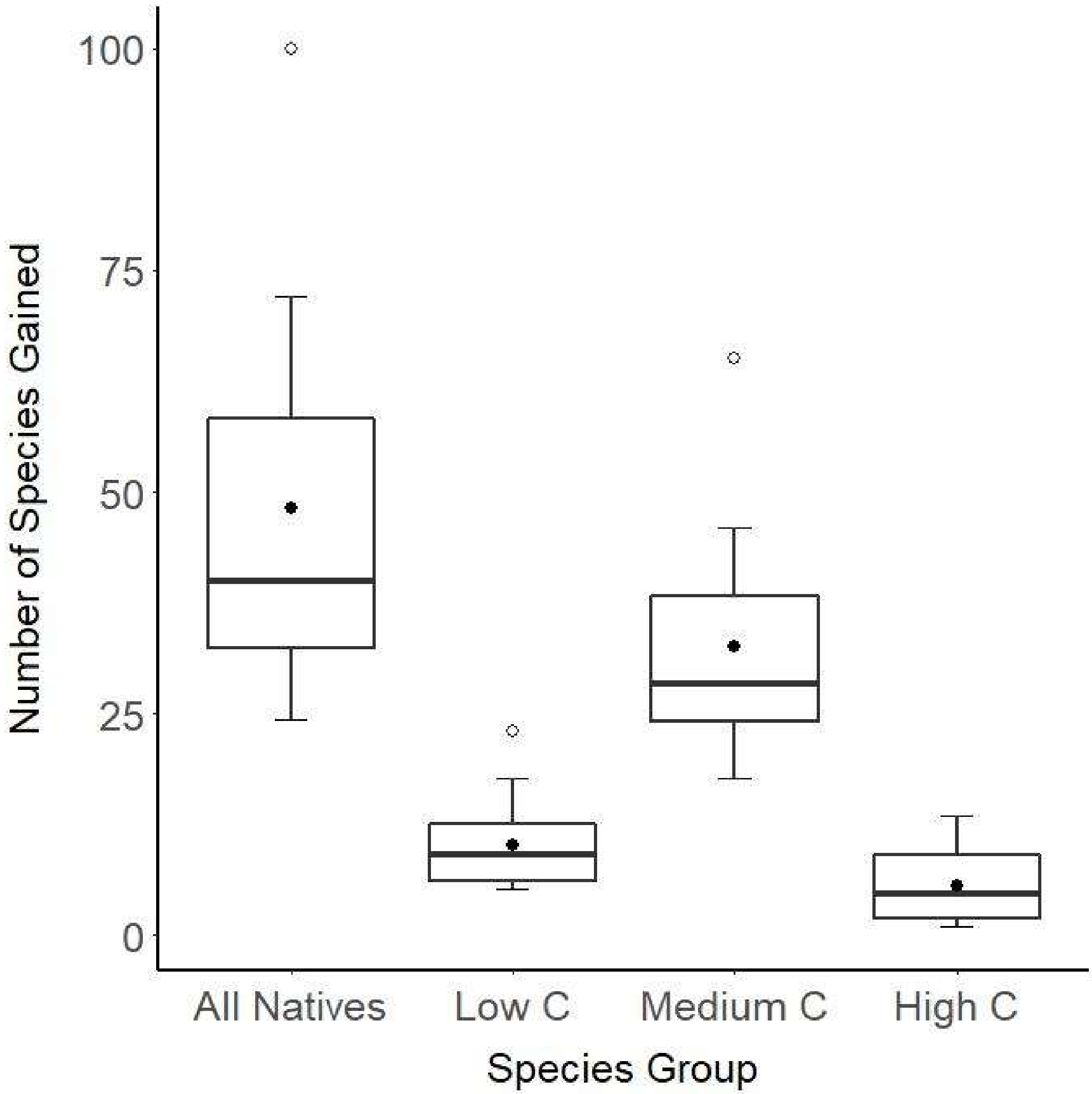

Figure 4.3. Box-and-whisker plots showing the average number of native plant species that were gained as a result of each wetland mitigation bank for all native species and across floristic quality groups of plant species. Box-and-whisker plots show the distribution of average number of species gained from 13 banks, with the value for each bank being the average of 1,000 trials for that bank. The floristic quality groups contain the following range of C-values: 0-2 in Low C, 3-7 in Medium C, and 8-10 in High C. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), the distribution of data points (whiskers), and outliers (open circles). All trials were run using a mitigation ratio of 1.5:1.

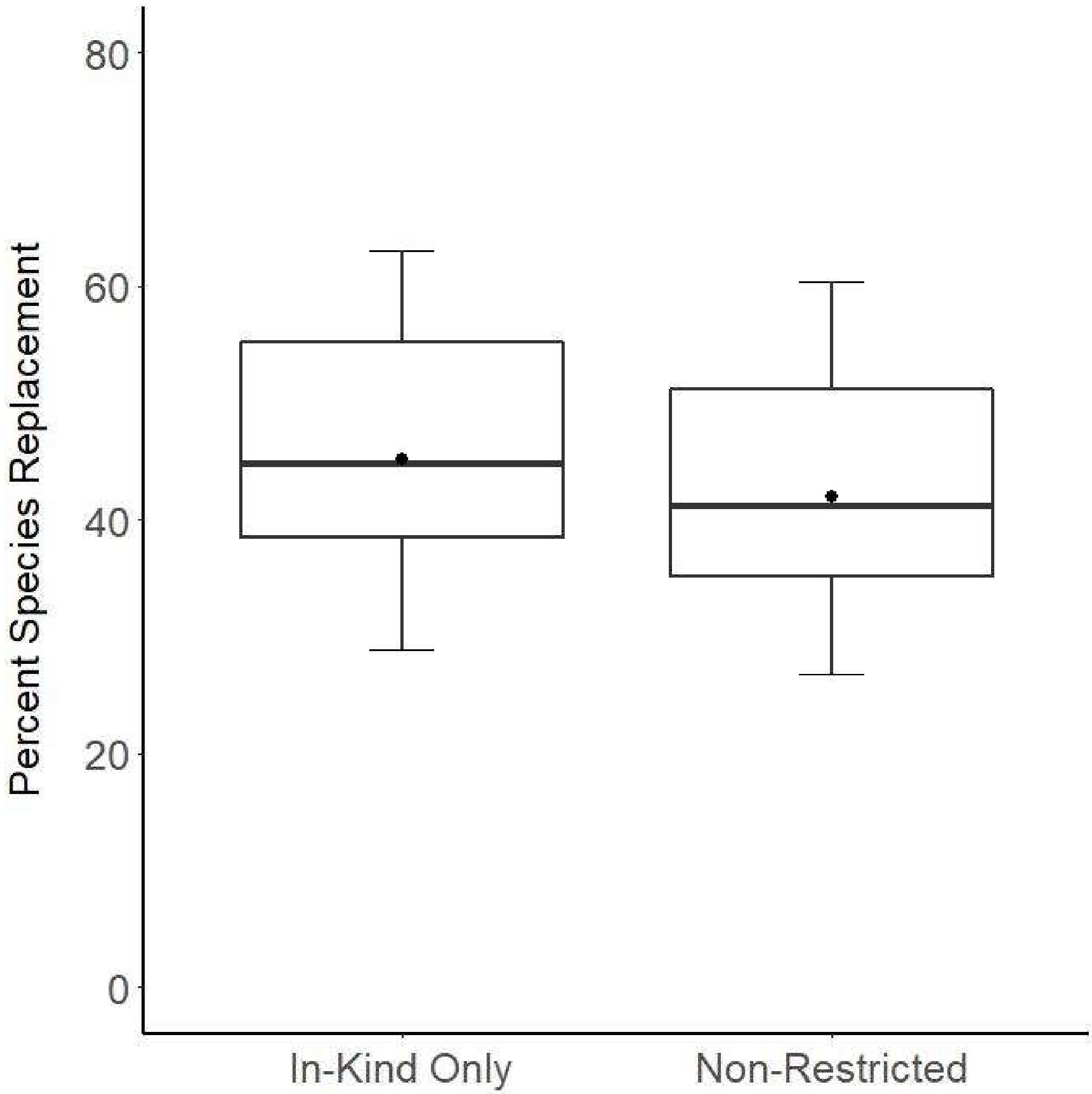

Figure 4.4. Box-and-whisker plots showing the average percentage of native plant species that were replaced at each wetland mitigation bank under in-kind only and non-restricted mitigation conditions. In-kind only trials included only natural wetlands which were classified as an open herbaceous plant community type (846 sites). Non-restricted trials included all natural wetland sites (1,530 sites). Box-and-whisker plots for each treatment group show the distribution of average replacement values from 13 banks, with the value for each bank being the average of 1,000 trials for that bank. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), and the distribution of data points (whiskers).

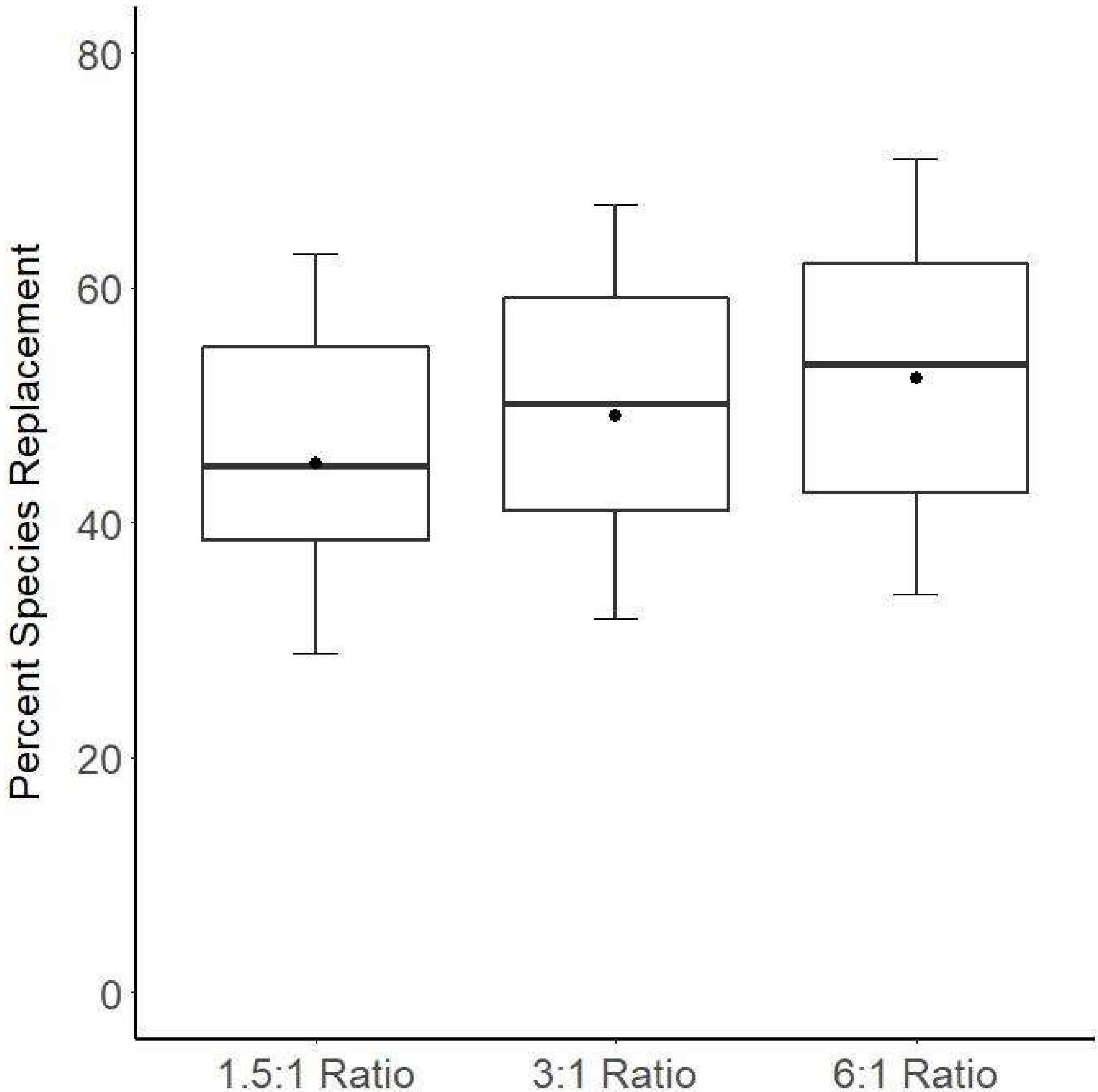

Figure 4.5. Box-and-whisker plots showing the average percentage of native plant species that were replaced at each wetland mitigation bank at different mitigation ratios. Box-and-whisker plots for each treatment group show the distribution of average replacement values from 13 banks, with the value for each bank being the average of 1,000 trials for that bank. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), and the distribution of data points (whiskers).

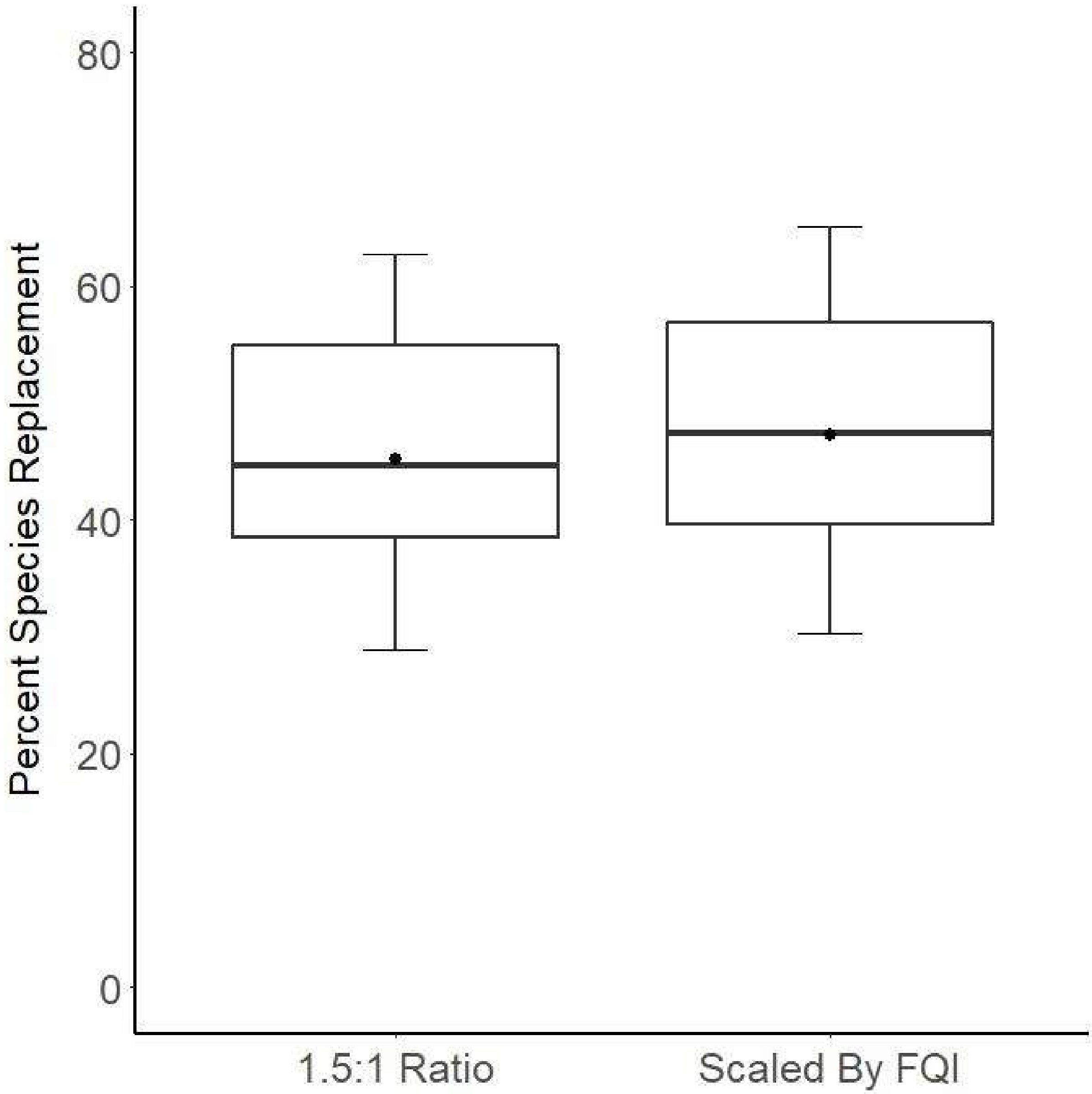

Figure 4.6. Box-and-whisker plots showing the average percentage of native plant species that were replaced at each wetland mitigation bank. Plots compare the results obtained using the base model ("1.5:1 Ratio") to those obtained using a mitigation ratio that was scaled by native FQI. Box-and-whisker plots for each treatment group show the distribution of average replacement values from 13 banks, with the value for each bank being the average of 1,000 trials for that bank. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), and the distribution of data points (whiskers).

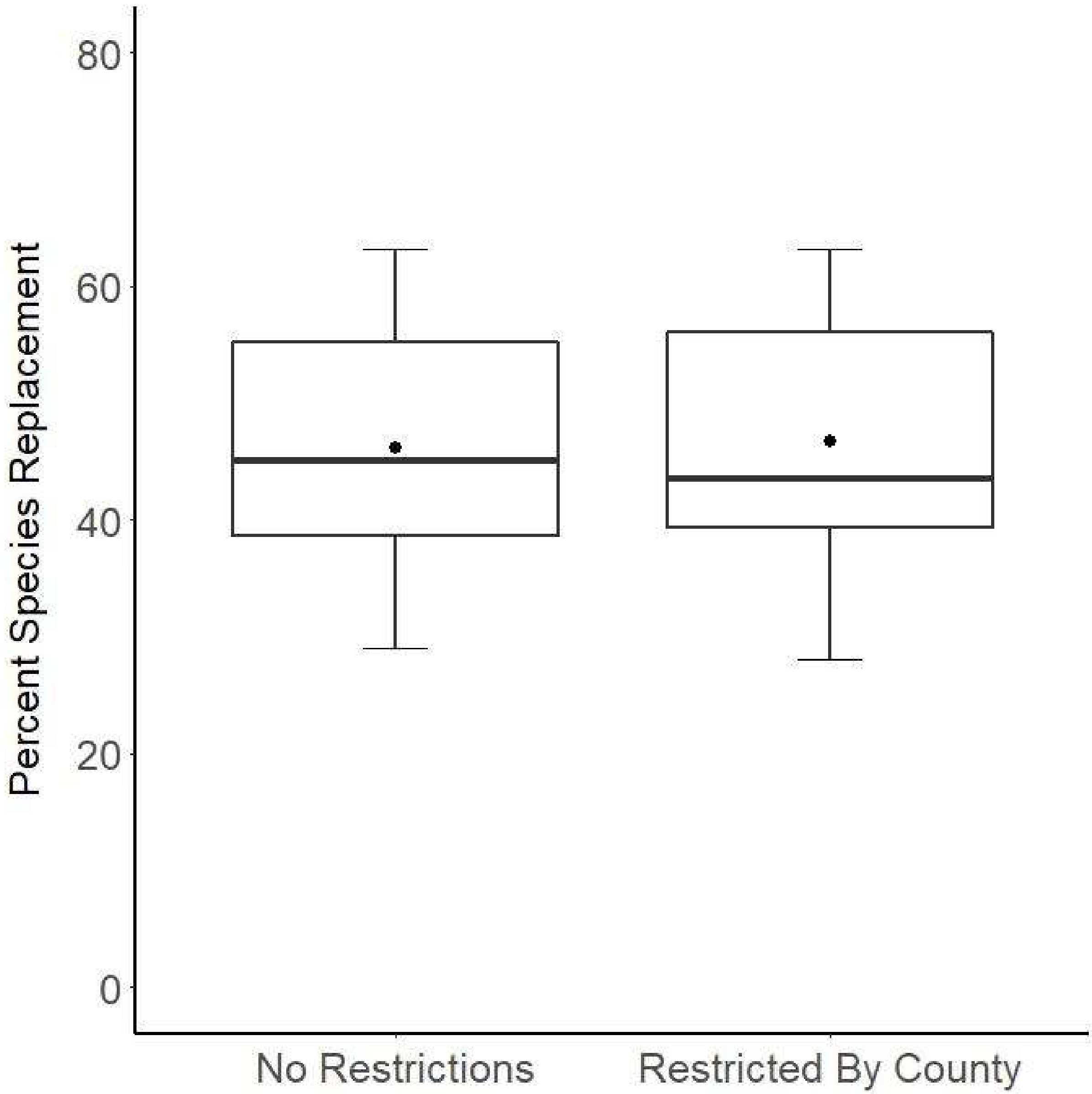

Figure 4.7. Box-and-whisker plots showing the average percentage of native plant species that were replaced at each wetland mitigation bank. Plots compare the results obtained using the base model with no geographic restrictions to a model run in which credit transactions were restricted to banks and natural wetlands that occurred within the same county. Box-and-whisker plots for each treatment group show the distribution of average replacement values from 12 banks, with the value for each bank being the average of 1,000 trials for that bank. Box-and-whisker plots show the mean (dark circles), median (horizontal line in box), interquartile range (entire box), and the distribution of data points (whiskers). All trials were run using a mitigation ratio of 1.5:1.

4.6 Tables

Table 4.1. Sample means and standard deviations for the average number of native plant species that were replaced, lost, and gained, and for the percentage of species that were replaced, by a single wetland mitigation bank during a run of the simulation base model. The values provided were calculated across 13 banks and 1,000 trials at each bank. The floristic quality groups contain the following range of C-values: 0-2 in Low C, 3-7 in Medium C, and 8-10 in High C. All trials were run using a mitigation ratio of 1.5:1.

	All Natives		Low C		Medium C		High C	
	Mean	*SD*	Mean	*SD*	Mean	*SD*	Mean	*SD*
Number of Species Replaced	68.4	18.6	28.1	9.8	39.5	9.4	0.8	0.6
Number of Species Lost	85.2	21.8	28.0	9.6	50.3	12.6	6.9	2.0
Number of Species Gained	48.2	21.6	10.2	5.5	32.5	13.4	5.5	4.4
Percent Species Replacement	45.2	10.3	50.4	16.1	44.8	8.7	12.3	8.1

Table 4.2. The 25 plant species that were replaced (present both in destoryed natural wetlands and in banks) most frequently as the result of the model simulating wetland mitigation bank transactions. The given values indicate the percentage of trials in which a species experienced each possible outcome of the simulation, and are sorted by the percentage replaced. These values were averaged across 13 banks and 1,000 trials at each bank. All trials were run using a mitigation ratio of 1.5:1.

Species	C-Value	Growth Form	Lifespan	Outcome of Simulations (Percent Replacement)			
				Replaced	Lost	Gained	Not Present
Verbena hastata	4	Forb	Perennial	99.3	0.0	0.7	0.0
Scirpus atrovirens	4	Sedge	Perennial	98.9	0.0	1.1	0.0
Juncus dudleyi	4	Forb	Perennial	98.5	0.0	1.5	0.0
Ambrosia trifida	0	Forb	Annual	98.1	0.0	1.9	0.0
Asclepias incarnata	4	Forb	Perennial	97.9	0.0	2.1	0.0
Carex vulpinoidea	2	Sedge	Perennial	97.9	0.0	2.1	0.0
Bidens frondosa	1	Forb	Annual	97.7	0.0	2.3	0.0
Helianthus grosseserratus	2	Forb	Perennial	97.2	0.0	2.8	0.0
Juncus torreyi	4	Forb	Perennial	95.7	0.0	4.3	0.0
Aster simplex	3	Forb	Perennial	92.1	7.7	0.2	0.0
Eleocharis erythropoda	2	Sedge	Perennial	92.0	7.7	0.3	0.0
Scirpus validus var. *creber*	5	Sedge	Perennial	91.2	7.7	1.1	0.0
Leersia oryzoides	4	Grass	Perennial	90.5	6.8	1.8	0.9
Echinochloa crusgalli	0	Grass	Annual	90.0	6.8	2.3	0.9
Ambrosia artemisiifolia var. *elatior*	0	Forb	Annual	89.6	7.5	2.7	0.2
Epilobium coloratum	3	Forb	Perennial	88.4	6.2	3.9	1.5
Solidago graminifolia	4	Forb	Perennial	88.2	7.6	4.1	0.1
Polygonum pensylvanicum	0	Forb	Annual	87.1	7.2	5.2	0.5
Salix interior	1	Shrub	Perennial	84.4	15.3	0.2	0.0
Aster novae-angliae	4	Forb	Perennial	84.2	7.3	8.2	0.4
Alisma subcordatum	4	Forb	Perennial	83.9	14.9	0.7	0.5
Typha latifolia	1	Forb	Perennial	83.9	14.8	0.7	0.6
Solidago gigantea	4	Forb	Perennial	83.5	15.2	1.2	0.2
Sagittaria latifolia	4	Forb	Perennial	82.5	6.6	9.8	1.1
Scirpus fluviatilis	4	Sedge	Perennial	76.4	22.0	0.6	1.0

Table 4.3. The 25 plant species that were lost (present in destroyed natural wetlands but not in banks) most frequently as the result of the model simulating wetland mitigation bank transactions. The given values indicate the percentage of trials in which a species experienced each possible outcome of the simulation, and are sorted by the percentage lost. These values were averaged across 13 banks and 1,000 trials at each bank. All trials were run using a mitigation ratio of 1.5:1.

Species	C-Value	Growth Form	Lifespan	Outcome of Simulations (Percent Replacement)			
				Replaced	Lost	Gained	Not Present
Sambucus canadensis	1	Shrub	Perennial	15.3	83.3	0.1	1.3
Parthenocissus quinquefolia	2	Vine	Perennial	15.0	81.7	0.4	3.0
Urtica procera	2	Forb	Perennial	15.2	79.9	0.2	4.7
Salix amygdaloides	5	Tree	Perennial	22.1	75.8	0.9	1.1
Cornus obliqua	6	Shrub	Perennial	22.5	75.2	0.6	1.8
Ulmus americana	3	Tree	Perennial	22.3	73.5	0.7	3.4
Equisetum arvense	0	Fern	Perennial	20.7	72.7	2.4	4.2
Ribes americanum	7	Shrub	Perennial	0.0	72.5	0.0	27.5
Polygonum punctatum	6	Forb	Annual	6.2	72.4	1.5	19.9
Polygonum scandens	1	Vine	Perennial	0.0	70.1	0.0	29.9
Solidago canadensis	1	Forb	Perennial	30.7	69.1	0.0	0.1
Fraxinus pennsylvanica var. *subintegerrima*	1	Tree	Perennial	30.4	68.9	0.4	0.3
Impatiens capensis	3	Forb	Annual	29.4	68.0	1.3	1.2
Carex pellita	4	Sedge	Perennial	25.9	63.3	4.8	6.0
Lippia lanceolata	6	Forb	Perennial	5.4	63.1	2.3	29.2
Ceratophyllum demersum	5	Forb	Perennial	0.0	60.7	0.0	39.3
Apocynum cannabinum	4	Forb	Perennial	37.1	60.5	1.4	1.1
Potamogeton nodosus	7	Forb	Perennial	10.2	57.9	5.2	26.7
Rhus radicans	2	Vine	Perennial	23.7	57.7	7.1	11.6
Erechtites hieracifolia	2	Forb	Annual	25.2	57.1	5.6	12.1
Acnida altissima	0	Forb	Annual	27.6	56.8	3.1	12.4
Geum laciniatum var. *trichocarpum*	2	Forb	Perennial	35.5	56.5	3.0	5.0
Galium aparine	1	Forb	Annual	24.7	55.5	6.0	13.8
Pilea pumila	5	Forb	Annual	16.8	55.4	6.3	21.5
Veronica peregrina	0	Forb	Annual	26.9	55.4	3.9	13.8

Table 4.4. The 25 plant species that were gained (absent from destoryed natural wetlands but present in banks) most frequently as the result of the model simulating wetland mitigation bank transactions. The given values indicate the percentage of trials in which species experienced each possible outcome of the simulation, and are sorted by the percentage gained. These values were averaged across 13 banks and 1,000 trials at each bank. All trials were run using a mitigation ratio of 1.5:1.

Species	C-Value	Growth Form	Lifespan	Outcome of Simulations (Percent Replacement)			
				Replaced	Lost	Gained	Not Present
Carex scoparia	7	Sedge	Perennial	3.2	0.3	89.1	7.3
Zizia aurea	7	Forb	Perennial	12.4	0.8	79.9	6.9
Elymus canadensis	4	Grass	Perennial	6.0	1.0	78.6	14.4
Rudbeckia hirta	1	Forb	Perennial	27.0	0.0	73.0	0.0
Ratibida pinnata	4	Forb	Perennial	4.6	1.5	72.3	21.6
Silphium integrifolium	5	Forb	Perennial	6.8	3.5	62.4	27.3
Andropogon gerardii	5	Grass	Perennial	33.8	3.3	58.5	4.4
Carex bebbii	6	Sedge	Perennial	12.0	4.7	57.2	26.1
Coreopsis tripteris	5	Forb	Perennial	0.0	0.0	53.8	46.2
Physostegia virginiana	6	Forb	Perennial	15.7	5.3	53.5	25.4
Solidago altissima	1	Forb	Perennial	1.8	1.7	52.1	44.5
Silphium perfoliatum	5	Forb	Perennial	48.4	0.0	51.6	0.0
Sorghastrum nutans	5	Grass	Perennial	25.5	6.9	51.5	16.1
Juncus effusus	7	Forb	Perennial	41.7	4.5	50.6	3.2
Helenium autumnale	5	Forb	Perennial	39.1	7.2	45.5	8.2
Elymus virginicus	4	Grass	Perennial	46.9	3.3	45.4	4.4
Physalis virginiana	4	Forb	Perennial	1.7	2.0	44.5	51.8
Potentilla norvegica	0	Forb	Annual	48.3	3.4	44.0	4.3
Lepidium virginicum	0	Forb	Annual	3.6	2.0	42.5	51.8
Solidago riddellii	7	Forb	Perennial	4.7	5.4	41.5	48.5
Polygonum hydropiperoides	7	Forb	Perennial	7.0	8.9	39.2	44.9
Scirpus cyperinus	6	Sedge	Perennial	45.9	6.8	38.7	8.6
Agalinis tenuifolia	7	Forb	Annual	7.5	8.8	38.7	45.0
Bidens cernua	5	Forb	Annual	53.7	2.5	38.6	5.2
Heliopsis helianthoides	5	Forb	Perennial	1.2	2.0	37.3	59.5

Table 4.5. Trial information and parameter estimates for percent species replacement from tests of different policy conditions tested using the model. Parameter estimates for species replacement were obtained using linear mixed modelling. The number of bank sites tested for the trials of in-kind only vs non-restricted mitigation, mitigation ratio, and ratio scaled by native FQI was 13 and the number of bank sites tested for the trial restricting transactions by county was 12.

		Impacted Natural Wetlands		Estimated Percent Species Replacement	
Trial Type	Treatment Level	Mean Number of Sites	Mean Number of Acres	Mean	95% C. I.
In-Kind Only vs. Non-Restricted	In-Kind Only	29.8	29.9	45.2	(39.4, 50.9)
	Non-Restricted	29.5	29.9	42.0	(35.9, 48.1)
Mitigation Ratio	1.5:1	30.0	29.9	45.1	(38.9, 51.3)
	3:1	17.3	15.0	49.1	(42.0, 56.2)
	6:1	10.9	7.6	52.3	(45.2, 59.5)
Scaled By Native FQI	Base Model	29.9	29.9	45.1	(39.2, 51.0)
	Scaled By Native FQI	24.6	21.6	47.2	(40.9, 53.5)
Restricted By County	Base Model	28.9	28.8	46.2	(40.0, 52.4)
	Restricted By County	29.9	28.8	46.8	(39.2, 54.4)

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CHAPTER 5: SUMMARY

5.1 Summary

In this thesis I have presented three studies investigating the compliance and ecological outcomes of wetland mitigation banks. To complete these studies, I used data collected from banks operated within the Chicago District of the U.S. Army Corps of Engineers and from natural wetlands in the region.

In my first study (Chapter 2), I examined regulatory compliance in mitigation banks, and my research objectives were to determine 1) how effectively banks met their regulatory performance standards during their final year of mandatory monitoring and 2) if the scores of the vegetation metrics used for performance standards changed over time in banks during their monitoring periods. I found that most of the banks in my study were not able to meet all their performance standards, especially standards limiting dominance by invasive species, though banks were much more successful at exceeding minimum thresholds for native species richness and dominance. I found some evidence that native perennial species richness and measures of floristic quality increased in banks during their monitoring periods.

In my second study (Chapter 3), I collected field vegetation data from banks and used several vegetation-based metrics and measures of community composition to compare the plant communities in wetland mitigation banks to those in natural wetlands representing a gradient of degradedness and ecological quality. My objectives were to 1) compare banks to natural wetlands of variable quality using vegetation metrics, 2) determine if the values for these vegetation indicators measured in banks were related to bank age, and 3) compare the plant community composition in banks to that in natural wetlands. My results, both for univariate vegetation metrics and community composition, gave clear evidence that banks possess wetland plant communities of greater ecological quality than low-quality, degraded natural wetlands, but that banks are not close to reaching equivalence with high-quality reference natural wetlands. The plant communities in banks do seem to be distinct from all natural wetlands, a condition that may be driven in emergent wetlands in banks by the abundance of the non-native species *Typha angustifolia* and *Phragmites australis*. I found some evidence that dominance by native species may be lower in older banks, but otherwise did not find evidence for a relationship between vegetation metrics and bank age.

In my third study (Chapter 4), I used existing plant species lists from banks and natural wetlands and developed a unique simulation modeling approach to determine how effectively banks may be able to replace the specific plant species that are lost from the natural wetlands for which banks may be used as compensation. I sought to 1) determine what percentage of the native plant species present in impacted natural wetlands banks may typically be able to replace, 2) assess how banks' ability to replace plant species varies by species floristic quality and for individual species, and 3) test if changes to certain policy conditions may result in an increase in species replacement by banks. I found that, under regulatory conditions that are typical for the Chicago District of the Corps, banks were able to replace, on average, about 45% of the native plant species present in the natural wetlands which used banks as compensation. Banks were much more effective at replacing species with high to moderate tolerance for human disturbance than for species with a high affinity to undisturbed natural communities. I found evidence that increasing the mitigation ratio governing credit transactions may result in moderate increases in species replacement, but that banks still seemed unable to replace many of the species present in impacted natural wetlands.

Negative impacts to wetlands in the United States are allowed by the Army Corps because of the assumption that wetland mitigation, including wetland mitigation banks, can be used to effectively compensate for those impacts so that no net loss of wetland resources occurs (Corps and EPA 1990). As wetland mitigation banking has been prioritized by the Corps (Corps and EPA 2008) and become used more frequently (IWR 2015, Hough and Harrington 2019) it has become important to evaluate this assumption by determining if wetland banks are able to reach equivalence with natural wetlands and replace the specific wetland resources found within them. This thesis has addressed this research challenge by providing new methods and data with which to assess the outcome of wetland mitigation banking.

5.2 References

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